

# FINAL REPORT

## Indicators of Ecological Change

SERDP Project RC-1114C

MARCH 2005

Virginia H. Dale  
Oak Ridge National Laboratory

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Final Report on

**Indicators of Ecological Change**

*CS 1114C*

A research effort supported by the

Strategic Environmental Research and  
Development Program (SERDP)  
Ecosystem Management Program (SEMP)

Principal Investigator: Dr. Virginia H. Dale  
Oak Ridge National Laboratory  
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**Executive Summary for *Indicators of Ecological Change***  
**SERDP Ecosystem Management Project CS 1114C**  
**Principal Investigator: Virginia H. Dale**  
**March 2005**

**1. Background**

Some of the finest surviving natural habitat in the United States is on military reservations where land has been protected from development. However, military training activities often necessitate ecological disturbance to that habitat. Fort Benning, Georgia, contains active infantry training grounds and more than 65,000 ha of soils capable of supporting longleaf pine (*Pinus palustris*) forest, a greatly reduced forest type in the North America. As longleaf pine forests are the primary habitat for the federally-endangered red-cockaded woodpecker (*Picoides borealis*), land managers at this installation have a dual charge both to maintain conditions for mechanized training activities and to conserve the integrity of this landscape.

Characterizing how resource use and management activities affect ecological conditions is necessary to document and understand ecological changes. Resource managers on military installations have the delicate task of balancing the need to train soldiers effectively with the need to maintain ecological integrity. Ecological indicators can play an important role in the management process by providing feedback on the impacts that training has on environmental characteristics,

The challenge in using ecological indicators is determining which of the numerous measures of ecological systems best characterize the entire system but are simple enough to be effectively monitored and modeled. Ecological indicators quantify the magnitude of stress, degree of exposure to stress, or degree of ecological response to the exposure and are intended to offer a simple and efficient method to examine ecological composition, structure, and function of whole systems. The use of ecological indicators as a monitoring device relies on the assumption that the presence or absence of, and fluctuations in, these indicators reflect changes taking place at various levels in the ecological hierarchy.

Although few scientists deny the benefits that indicators provide to research and management efforts, three concerns jeopardize the use of ecological indicators as a management tool.

- **Management and monitoring programs often depend on a small number of indicators and, as a consequence, fail to consider the full complexity of the ecological system.** By selecting only one or a few indicators, the focus of the ecological management program becomes narrow, and an oversimplified understanding of the spatial and temporal interactions is created. This simplification often leads to poorly informed management decisions. Indicators should be selected from multiple levels in the ecological hierarchy in order to effectively monitor the multiple levels of complexity within an ecological system.
- **Choice of ecological indicators is often confounded by management programs that have vague management goals and objectives.** Unclear or ambivalent goals and objectives can lead to “the wrong variables being measured in the wrong place at the wrong time with poor precision or reliability” (Noss and Cooperrider 1994). Primary goals and objectives should be determined early in



the process in order to focus management. Ecological indicators can then be selected from system characteristics that most closely relate to those management concerns.

- **Management and monitoring programs often lack scientific rigor because of their failure to use a defined protocol for identifying ecological indicators.**

Lack of a procedure for selecting ecological indicators makes it difficult to validate the information provided by those indicators. Until a standard method is established for selecting and using indicators, interpretation of their change remains speculative. The creation and use of a standard procedure for the selection of ecological indicators allow repeatability, avoid bias, and impose discipline upon the selection process, ensuring that the selection of ecological indicators encompasses management concerns.

Development of a procedure for ecological indicator selection that is based on a hierarchical framework and grounded in clear management goals will address concerns associated with the subjective and disorganized methods often used. We present such an approach for identifying ecological indicators. The ultimate goal is to establish the use of ecological indicators as a means for including ecological objectives and concerns in management decisions.

The approach is applied to Department of Defense (DoD) lands in the United States (US) where military land contributes significantly to habitat conservation. The DoD manages more than 10 million ha representing more than 450 installations nationally. Although this area is much less land than the area managed by the Department of the Interior (180 million ha) or the United States Forest Service (USFS) (77 million ha), greater species diversity per unit area exists within DoD lands than within lands of any other federal ownership (except Department of Energy lands). In addition, DoD lands contain more endangered species per unit area than any other federal land management agency, and individual installations often contain more land than most national parks or wildlife refuges. While a portion of all military installations is highly disturbed, most land within military bases is designated as training areas or buffer zones and, therefore, remains in a relatively natural state, providing numerous habitats and a haven for associated species. These facts coupled with the DoD's commitment to ecosystem management and conservation provide an outstanding opportunity for establishing sustainable management practices that ensure the future of these habitat and species resources. Although its mission is military training and testing, the DoD recognizes the relationship between its military mission and the natural resources upon which that mission depends, and, therefore, the benefits of creating and implementing long-term ecosystem-management plans (1994 Memorandum from the Office of Deputy Secretary of Defense).

This research explored the use of ecological indicators as a land management tool, focusing on the development of a procedure for selecting and monitoring ecological indicators. In response to the limitations that currently hamper the effectiveness of ecological indicators as a management device, we considered a hierarchical approach to land management and the role indicators can play in providing the monitoring information required by ecosystem management. This summary discusses criteria and presents the suite of indicators that we considered for military land use at the landscape, watershed and plot level. The development and implementation of land-management



strategies for military land not only provide valuable tools for the continued mission of the DoD but also suggest how ecological indicators can be used for ecosystem management of other multiple-use lands.

## 2. Criteria for selecting ecological indicators

Selection of effective indicators is key to the overall success of any monitoring program. In general, ecological indicators need to capture the complexities of the ecosystem yet remain simple enough to be easily and routinely monitored. In order to define ecological indicators, however, it is first necessary to set forth criteria used to select potential ecological indicators. Building upon discussions in the scientific literature and discussions with the other SERDP Environmental Management Project (SEMP) research teams and the resource managers at Fort Benning, we suggest that ecological indicators should meet the following criteria:

- **Be easily measured.** The indicator should be straightforward and relatively inexpensive to measure. The metric needs to be easy to understand, simple to apply, and provide information to managers and policymakers that is relevant, scientifically sound, easily documented, and cost-effective.
- **Be sensitive to stresses on the system.** The ideal ecological indicator is responsive to stresses placed on the system by human actions while also having limited and documented sensitivity to natural variation. While some indicators may respond to all dramatic changes in the system, the most useful indicator is one that displays high sensitivity to a particular and, perhaps, subtle stress, thereby serving as an early indicator of reduced system integrity.
- **Respond to stress in a predictable manner.** The indicator response should be unambiguous and predictable even if the indicator responds to the stress by a gradual change. Ideally, there is some threshold response level at which the observable response occurs before the level of concern.
- **Be anticipatory, that is, signify an impending change in key characteristics of the ecological system.** Change in the indicator should be measurable before substantial change in ecological system integrity occurs.
- **Predict changes that can be averted by management actions.** The value of the indicator depends on its relationship to possible changes in management actions.
  - **Are integrative: the full suite of indicators provides a measure of coverage of the key gradients across the ecological systems (e.g., gradients across soils, vegetation types, temperature, space, time, etc.).** The full suite of indicators for a site should integrate across key environmental gradients. For example, no single indicator is applicable across all spatial scales of concern. The ability of the suite of indicators to embody the diversity in soils, topography, disturbance regimes, and other environmental gradients at a site should be considered.
- **Have a known response to disturbances, anthropogenic stresses, and changes over time.** The indicator should have a well documented reaction to both natural disturbance and to anthropogenic stresses in the system.
- **Have low variability in response.** Indicators that have a small range in response to particular stresses allow for changes in the response value to be better distinguished from background variability.



### 3. Landscape Indicators

This research examined landscape indicators that signal ecological change in both intensely used and lightly used lands at Fort Benning, Georgia. Changes in patterns of land cover through time affect the ecological system by altering the proportion and distribution of habitats for species that these cover types support. Landscape patterns, therefore, are important indicators of land-use impacts, past and present, upon the landscape. This analysis of landscape pattern began with a landscape characterization based on witness tree data from 1827 and the 1830s and remotely sensed data from 1974, 1983, 1991, and 1999. The data from the early 1800s, although coarse, were useful in characterizing the historical range of variability in ecological conditions for the area. The steps for the analysis involved the creation of a land-cover database and a time series of land cover maps, computation of landscape metrics, and evaluation of changes in those metrics over time as evidenced in the land-cover maps. We focused on five cover types (bare/developed land, deciduous forest, mixed forest, pine forest, and nonforest vegetated land), for they reveal information important to resources management at Fort Benning. An examination of land-cover class and landscape metrics, computed from the maps, indicated that a suite of metrics adequately describes the changing landscape at Fort Benning, Georgia. The most appropriate metrics were percent cover, total edge (km), number of patches, descriptors of patch area, nearest neighbor distance, the mean perimeter-to-area ratio, shape range, and clumpiness. Identification of such ecological indicators is an important component of building an effective environmental monitoring system.

### 4. Watershed Indicators

To evaluate watershed scale indicators of disturbance we studied twelve 2<sup>nd</sup>- and 3<sup>rd</sup>-order streams in the eastern part of the Fort Benning Military Installation (FBMI) that drained watersheds with a wide range of disturbance levels. We quantified watershed disturbance as the sum of the proportion of bare ground on slopes >3% and unpaved road cover within each watershed. Study streams drained watersheds ranging in disturbance from about ~2 to 14%. We then compared a variety of stream physical, chemical, and biological characteristics across this disturbance gradient to evaluate their usefulness as disturbance indicators.

We found that a number of stream characteristics were good indicators of watershed-scale disturbance at FBMI. Stream channel organic variables (i.e., amount of benthic particulate organic matter [BPOM] and coarse woody debris [CWD]) were highly related to watershed disturbance as was the degree of hydrologic flashiness (quantified by 4-hour storm flow recession constants) and bed stability. Among the stream chemistry variables, the concentrations of total and inorganic suspended sediments during baseflow and storm periods were excellent indicators of disturbance, typically increasing with increasing disturbance levels. In addition, baseflow concentrations of dissolved organic carbon and soluble reactive phosphorus were good disturbance indicators, declining with increasing disturbance levels. Among biological variables, stream benthic macroinvertebrates also were good indicators of watershed-scale disturbance. Traditional measures such as community richness (e.g., number of Ephemeroptera, Plecoptera, and Trichoptera [EPT] taxa and richness of Chironomidae) negatively corresponded with watershed disturbance; however, except for chironomid richness, all measures showed



high variation among seasons and annually. A multimetric index previously designed for Georgia streams (Georgia Stream Condition Index [GASCI]) consistently indicated watershed disturbance and also showed low seasonal and inter-annual variation. Low diversity of fish precluded use of traditional measures (i.e., richness, diversity), however the proportional abundance of the two dominant populations (*P. euryzonus* and *S. thoreauianus*) were strongly but oppositely associated with disturbance, with *P. euryzonus* and *S. thoreauianus* being negatively and positively related to disturbance, respectively. Finally, historical land use explained more variation in contemporary bed stability and longer-lived, low turnover taxa than contemporary land use, suggesting a “legacy” effect on these stream measures. Prior to identification and use of potential indicators, managers at FBMI should acknowledge historical land use and the possible presence of legacy effects on aquatic physicochemical and biotic conditions.

## 5. Plot-level indicators

### A. Vegetation indicators

Environmental indicators for longleaf pine (*Pinus palustris*) ecosystems need to include some measure of understory vegetation because of its responsiveness to disturbance and management practices. To examine the characteristics of understory species that distinguish between disturbances induced by military traffic, we randomly established transects in four training intensity categories (reference = no military use, light = foot traffic only, moderate = marginal tracked vehicle use, and heavy = regular tracked vehicle use) and in an area that had been remediated following intense disturbance at Fort Benning, Georgia. A total of 137 plant species occurred in these transects with the highest diversity (95 species) in light training areas and the lowest (16 species) in heavily disturbed plots. Forty-seven species were observed in only one of the five disturbance categories. The variability in understory vegetation cover among disturbance types was trimodal ranging from less than 5% cover for heavily disturbed areas to 67% cover for reference, light, and remediated areas. High variability in species diversity and lack of distinctiveness of understory cover led us to consider Raunkiaer life form and plant families as indicators of military disturbance. Life form successfully distinguished between plots based on military disturbances. Species that are phanerophytes (trees and shrubs) were the most frequent life form encountered in sites that experienced light infantry training. Therophytes (annuals) were the least common life form in reference and light training areas. Chamaephytes (plants with their buds slightly above ground) were the least frequent life form in moderate and remediation sites. Heavy training sites supported no chamaephytes or hemicryptophytes (plants with dormant buds at ground level). The heavy, moderate, remediated, and reference sites were all dominated by cryptophytes (plants with underground buds) possibly because of their ability to withstand both military disturbance and ground fires (the natural disturbance of longleaf pine forests). Analysis of soils collected from each transect revealed that depth of the A layer of soil was significantly higher in reference and light training areas which may explain the life form distributions. In addition, the diversity of plant families and, in particular, the presence of grasses and composites were indicative of training and remediation history. These results are supported by prior analysis of life form distribution subsequent to other disturbances and demonstrate the ability of life form and plant families to distinguish between military disturbances in longleaf pine forests.



We further investigated the hypothesis that effects of military activity on these forests may be quantified by grouping understory species into life-forms by experimentally manipulating a longleaf pine forest using a mechanized vehicle. In May 2003, a D7 bulldozer removed extant vegetation and surface soil organic matter along three treatment transects. Braun-Blanquet vegetation surveys were recorded in June and September 2003 and 2004. Repeated measures analysis of variance was utilized to compare the response of 30 plots within the treatment transects to 30 plots in adjacent control transects. Total understory cover in the treatment transects decreased substantially in June, but rebounded by September 2003. Phanerophytes (trees and shrubs) in the treatment plots maintained reduced cover throughout the growing season. These findings support the use of Raunkiaer functional types in indicating the response of longleaf pine forests to mechanized disturbance. This approach should lead to a readily accessible measure of disturbance that can be assessed throughout the installation by land managers.

### ***B. Microbial indicators***

This research demonstrated that the soil microbial community of a longleaf pine ecosystem at Fort Benning, Georgia also responds to military traffic disturbances. Using the soil microbial biomass and community composition as ecological indicators, reproducible changes showed increasing traffic disturbance decreases soil viable biomass, biomarkers for microeukaryotes and Gram-negative bacteria, while increasing the proportions of aerobic Gram-positive bacterial and *actinomycete* biomarkers. Soil samples were obtained from four levels of military traffic (reference, light, moderate, and heavy) with an additional set of samples taken from previously damaged areas that were remediated via planting of trees and ground cover. Utilizing 17 phospholipid fatty acid (PLFA) variables that differed significantly with land usage, a linear discriminant analysis with cross-validation classified the four groups. Wilks' Lambda for the model was 0.032 ( $P < 0.001$ ). Overall, the correct classifications of profiles was 66% (compared to the chance that 25% would be correctly classified). Using this model, ten observations taken from the remediated transects were classified. One observation was classified as a reference, three as light trafficked, and six as moderately trafficked. Non-linear Artificial Neural Network (ANN) discriminant analysis was performed using the biomass estimates and all of the 61 PLFA variables. The resulting optimal ANN included five hidden nodes and resulted in an  $r^2$  of 0.97. The prediction rate of profiles for this model was again 66%, and the ten observations taken from the remediated transects were classified with four as reference (not impacted), two as moderate, and four as heavily trafficked. Although the ANN included more comprehensive data, it classified eight of the ten remediated transects at the usage extremes (reference or heavy traffic). Inspection of the novelty indexes from the prediction outputs showed that the input vectors from the remediated transects were very different from the data used to train the ANN. This difference suggests as a soil is remediated it does not escalate through states of succession in the same way as it descends following disturbance.

### ***C. Considering soil, vegetation and microbial indicators together***



Our results and those of Chuck Garten<sup>1</sup> (under another SEMP project) show that soil chemistry, soil microbes, and vegetation are all important indicators of ecological change. Accordingly, we questioned whether all of these indicators would be important if we combined these data into one analysis. Our hypothesis was that a suite of indicator types is necessary to explain ecological change. A discriminant function analysis was conducted to determine whether these ecological indicators could differentiate between different levels of military use. A combination of ten indicators explained 90% of the variation among plots from five different military-use levels. Results indicated that an appropriate suite of ecological indicators for military resource managers includes vegetation, microbial, and soil characteristics. This result is important for resource managers since many of the indicators are correlated, it implies that managers will have freedom to choose indicators that are relatively easy to measure, without sacrificing information.

## **6. Road and vehicle impacts at different scales at Fort Benning**

Roads and vehicles change the environmental conditions in which they occur. One way to categorize these effects is by the spatial scale of the cause and the impacts. Roads may be viewed from the perspective of road segments, the road network, or roads within land ownership or political boundaries such as counties. Our research examined the hypothesis that the observable impacts of roads on the environment depend on spatial resolution. To examine this hypothesis, the environmental impacts of vehicles and roads were considered at four scales in west central Georgia in and around Fort Benning: a second-order catchment, a third-order watershed, the entire military installation, and the five-county region including Fort Benning. Impacts from an experimental path made by a tracked vehicle were examined in the catchment. Land-cover changes discerned through remote sensing data over the past three decades were considered at the watershed and installation scales. A regional simulation model was used to project changes in land cover for the five-county region. Together these analyses provide a picture of the how environmental impacts of roads and vehicles can occur at different spatial scales. Following tracked vehicle impact with a D7 bulldozer, total vegetation cover responded quickly, but the plant species recovered differently. Soils were compacted in the top 10 cm and are likely to remain so for some time. Examining the watershed from 1974 to 1999 revealed that conversion from forest to nonforest was highest near unpaved roads and trails. At the installation scale, major roads as well as unpaved roads and trails were associated with most of the conversion from forest to nonforest. For the five-county region, most of the conversion from forest to nonforest is projected to be due to urban spread rather than direct road impacts [using a model developed for another SERDP project]. The study illustrates the value of examining the effects of roads at several scales of resolution and shows that road impacts in west central Georgia are most important at local to subregional scales.

## **7. Technology transfer**

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<sup>1</sup> Garten, C.T., T.L. Ashwood, and V.H. Dale. 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. *Ecological Indicators* 3(3) 171-180.



The objective of this study was to identify indicators that signal ecological change in intensely and lightly used ecological systems (all Fort Benning has had some anthropogenic changes) that could be used by the resource managers. Because the intent was that these indicators become a part of the ongoing monitoring system at the installation, the indicators were selected for their feasibility for the installation staff to measure and interpret. While the focus was on Fort Benning, the goal was to develop an approach to identify indicators that would be useful to a diversity of military installations and other land ownerships (in some cases the actual indicators may be adopted). The intent of this identification of indicators was to improve managers' ability to manage activities that are likely to be damaging and to prevent long-term, negative effects. Therefore, we examined a suite of variables needed to measure changes in ecological conditions. The results of our research were presented to the Fort Benning Resource managers in a half day workshop in February 2005.



## **Section 2: Objectives of the SEMP Project: *Indicators of Ecological Change***

The technical objective of this effort was to identify indicators that signal ecological change in intensely and lightly used ecological systems (all Fort Benning has had some anthropogenic changes). Because the intent was that these indicators become a part of the ongoing monitoring system at the installation, the indicators were selected that are feasible for the installation staff to measure and interpret. The focus was on Fort Benning, but the goal was to develop an approach to identify indicators that would be useful to a diversity of military installations (in some cases the actual indicators may be applied across many installations). Because some of these effects may be long-term or may occur after a lag time, early indications of both current and future change were identified. The intent of this identification of indicators was to improve managers' ability to manage activities that are likely to be damaging and to prevent long-term, negative effects. Therefore, a suite of variables is needed to measure changes in ecological conditions. The suite that we examined includes measures of terrestrial biological integrity, stream chemistry and aquatic biological integrity, and soil microorganisms as a measure of below-ground integrity of the ecosystem.

It is critical to be able to define the significant ecological changes within a system that naturally has disturbances, such as droughts or fires, or has other natural changes, such as succession, occurring. We also considered how human activities may or may not mimic the frequency and intensity of natural disturbances and how to predict responses outside the range of natural variation. Another issue considered was how these human impacts have implications on the ecological system both inside and outside the installation.

Fort Benning provides an excellent opportunity to examine potential indicators and may serve as a model of how such ecosystem studies can be done on military installations. Much data had already been collected there (as described in material available from Fort Benning) and a more comprehensive monitoring program was implemented during the research period. Fort Benning initiated the land condition trend analysis (LCTA), vegetation, and land-use monitoring survey in 1990. A great deal of information was available on geographic information systems (GIS). However, there was a strong need to further develop the monitoring program so that it meets the needs for ecosystem management at Fort Benning and so that monitoring is integrated into the decision-making process. Identification of indicators of ecological change is a critical part of this integration. Furthermore, with these indicators identified, it is hoped that the importance of the Fort Benning installation can be interpreted in view of the natural resources of the larger region by using the Natural Heritage databases of The Nature Conservancy. Our intent was to examine what additional system-level indicators should be added to current monitoring efforts, so that changes in ecosystem can be predicted. We also explored ways the land managers can use monitoring information to make decisions as part of the technology transfer part of the proposal.

The metrics examined represent the six principles of ecology that are needed to assure that fundamental processes of the Earth's ecosystems are sustained (as identified by the Land Use Committee of the Ecological Society of America). These ecological principles deal with time, species, place, productivity, disturbances, and the landscape.



Further aspects of our objectives are explained in the book chapter: Dale, V H., Mulholland, P., Olsen, L. M., Feminella, J., Maloney, K., White, D. C., Peacock, A., and Foster, T. 2004. "Selecting a Suite of Ecological Indicators for Resource Management," Pages 3-17 in *Landscape Ecology and Wildlife Habitat Evaluation: Critical Information for Ecological Risk Assessment, Land-Use Management Activities and Biodiversity Enhancement Practices*, **ASTM STP 11813**, L. A. Kapustka, H. Gilbraith, M. Luxon, and G. R. Biddinger, Eds., ASTM International, West Conshohocken, PA, 2004.



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## Selecting a Suite of Ecological Indicators for Resource Management

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**REFERENCE:** Dale, V. H., Mulholland, P. J., Olsen, L. M., Feminella, J. W., Maloney, K. O., White, D. C., Peacock, A., and Foster, T., "Selecting a Suite of Ecological Indicators for Resource Management," *Landscape Ecology and Wildlife Habitat Evaluation: Critical Information for Ecological Risk Assessment, Land-Use Management Activities and Biodiversity Enhancement Practices, ASTM STP 1458*, L. A. Kapustka, H. Galbraith, M. Luxon, and G. R. Biddinger, Eds., ASTM International, West Conshohocken, PA, 2004.

**ABSTRACT:** We discuss the use of ecological indicators as a natural resource management tool, focusing on the development and implementation of a procedure for selecting and monitoring indicators. Criteria and steps for the selection of ecological indicators are presented. The development and implementation of indicators useful for management are applied to Fort Benning, Georgia, where military training, controlled fires (to improve habitat for the endangered red cockaded woodpecker), and timber thinning are common management practices. A suite of indicators is examined that provides information about understory vegetation, soil microorganisms, landscape patterns, and stream chemistry and benthic macroinvertebrate populations and communities. For example, plants that are geophytes are the predominant life form in disturbed areas, and some understory species are more common in disturbed sites than in reference areas. The set of landscape metrics selected (based upon ability to measure changes through time or to differentiate between land cover classes) included percent cover, total edge (with border), number of patches, mean patch area, patch area range, coefficient of variation of patch area, perimeter/area ratio, Euclidean nearest neighbor distance, and clumpiness. Landscape metrics indicate that the forest area (particularly that of pine) has declined greatly since 1827, the date of our first estimates of land cover (based on witness tree data). Altered management practices in the 1990s may have resulted in further changes to the Fort Benning landscape. Storm sediment concentration profiles indicate that the more

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highly disturbed catchments had much greater rates of erosion and sediment transport to streams than less disturbed catchments. Disturbance also resulted in lower richness of EPT (i.e., number of taxa within the aquatic insect orders Ephemeroptera, Plecoptera, and Trichoptera) than in reference streams but similar total richness of invertebrate species. Each indicator provides information about the ecological system at different temporal and spatial scales.

**KEYWORDS:** disturbance, forests, indicators, resource management

## Introduction

The questions that our work addresses are on a local resource management level. What are the best indicators to be measuring? How can those metrics be properly interpreted? Because of its proactive mode of management, this effort focuses on lands owned and managed by the Department of Defense of the United States. We first examine criteria that are suitable for indicators and then consider steps of selection of indicators. A suite of indicators is proposed, and a case study dealing with potential indicators at Fort Benning, Georgia is presented. Overall, the paper provides insights into the value of indicators, how they are selected, and how they can be used.

## Criteria for Selecting Ecological Indicators

Criteria for selecting ecological indicators were developed based on the goal of capturing the complexities of the ecological system but remaining simple enough to be effectively and routinely monitored (Dale and Beyeler 2001):

- *Be easily measured.* The indicator should be easy to understand, simple to apply, and provide information that is relevant, scientifically sound, easily documented, and cost-effective (Lorenz et al. 1999).
- *Be sensitive to stresses of the system.* Ecological indicators should react to anthropogenic stresses placed on the ecological system, while also having limited and documented sensitivity to natural variation (Karr 1991).
- *Respond to stress in a predictable manner.* The response of the indicator should be decisive and predictable even if the indicator responds to the stress by a gradual change. Ideally, there is some threshold level at which the observed response is lower than the level of concern of the impact.
- *Be anticipatory: signify an impending change in key characteristics of the ecological system.* Change in the indicator should be measurable even before substantial change in the ecological system occurs.
- *Predict changes that can be averted by management actions.* The value of the indicator for management depends on its relationship to changes in human actions.
- *Be integrative: together with the full suite of indicators, provide a measure of coverage of the key gradients across the ecological systems (e.g., soils, vegetation types, temperature, etc.).* The full suite of indicators for a site should provide a synchronized perspective of the key attributes of major environmental gradients. These gradients may relate to time, space, soil properties, elevation, or any other factor that is important to the ecological system (e.g., see Figure 1).



- *Have a known response to natural disturbances, anthropogenic stresses, and ecological changes over time.* The indicator should have a definitive reaction to both natural disturbance and to anthropogenic stresses in the system. As ecological conditions change in a system (e.g., via succession), the response of the indicator should be predictable. This criterion most often pertains to metrics that have been extensively studied and have a clearly established pattern of response.
- *Have low variability in response.* Indicators that have a small range in response to particular stresses allow for change in the response value to be distinguished from background variability.

### Selecting Ecological Indicators

Identification of the key criteria for ecological indicators sets the stage for a seven-step procedure for selecting indicators. These steps are discussed in view of land use decisions on military lands but are applicable to resource issues on other public and private lands.

#### Hierarchical Overlap of Suite of Ecological Indicators Over Time

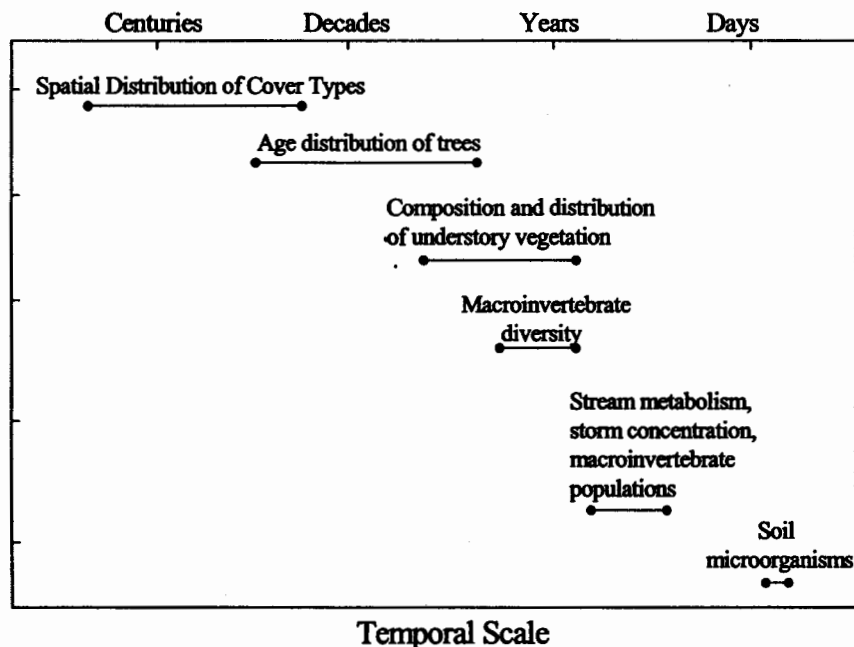


Figure 1 — A suite of indicators can be depicted across time



*Step 1: Identify Goals for the System.*

The first step in problem solving is to define the issue and develop clear goals and objectives. Often, goals are a compromise among the concerns of interested parties. Sometimes objectives change as adherence to one target compromises another. The more complex the nature of the problem, the more important it becomes to establish clear goals and objectives within the spatial and temporal parameters of the system. The selection of ecological indicators is complex in the sense that many factors are involved, feedbacks are common, and diverse groups of stakeholders have different perspectives, value systems, and intentions.

For spatial analysis, it is useful to consider both the immediate area of interest and a broader perspective. The area contained inside the socio-politically delineated boundary can be referred to as the *focal area*, for it is the area of immediate concern to the resource manager. In dealing with ecological management issues, situations often arise when it is useful to look outside of the focal area to a *context area*. Both the focal and context areas can be defined by ecological, social, or political concerns influencing system characteristics.

For the same reason that it is important to consider spatial context when assessing management options, it is also important to consider temporal context. Management areas are defined by past, present, and future social, political, and ecological influences. *Focal time* can be used to refer to the temporal context being considered in the focal area, and *context time* can be used to refer to the temporal context of the entire situation.

As an example, the focal area of conservation planning at Fort Benning is defined by the boundaries of the installation (a political unit), but the context area extends throughout much of the Southeast along the fall line that bisects Fort Benning and differentiates between the Coastal Plain and the Piedmont. One focal time for Fort Benning is the current time back to 1974 when the red cockaded woodpecker (*Picoides borealis*, RCW) was listed as an endangered species. Another focal time might be the last century, for Fort Benning has been the "home of the infantry" since 1918 and is now the site of major infantry and tank training exercises. The context time must consider the intensive agriculture practiced by European settlers since the 1800s and by Native Americans for centuries before that time (Kane and Keeton 1998; Foster et al. 2003). To better quantify the effects of agriculture before military activity began at Fort Benning, a vegetation map has been created based on witness tree surveys conducted in 1827 as part of land surveys performed in order to distribute the land (Olsen et al. 2001; Black et al. 2002; Foster et al. 2003). By viewing land use and land cover in the broad spatial and temporal context, meeting the management goals can be considered in light of these broader perspectives.

*Step 2: Identify Key Characteristics of the Ecological System*

Characteristics are the specific functional, compositional, and structural elements that, when combined, define the ecological system. All ecological systems have elements of composition and structure that arise through ecological processes. The characteristic



conditions of an area depend on sustaining key ecological functions that, in turn, produce additional compositional and structural elements. If the linkages between underlying processes, composition, and structural elements are broken, then sustainability is jeopardized and restoration may be difficult and complex.

Key characteristics include the physical features that allow species, ecosystems, or landscapes to occur. For example, at Fort Knox, Kentucky, locations of threatened calcareous habitats of rare species can be predicted based on a combination of soils, geology, and slope (Mann et al. 1999). This edaphic-based approach has also been used to identify locations of Henslow's sparrow (*Ammodramus henslowii*) habitat at Fort Knox and sites at Fort McCoy, Wisconsin, that can support wild lupine (*Lupinus perennis*), the sole host plant for the larvae of the endangered Karner blue butterfly (*Lycaecides melissa samuelis*) (Dale et al. 2000).

Identification of the key ecological characteristics of a system also involves attention to social, economic, and political features of a site. Combinations of social, economic, political, and ecological concerns, such as laws and regulations, peoples' values, regional economics, and ecological conditions, determine the importance of a characteristic. The Southern Appalachian Assessment (SAA) provides an example of multiple agencies working together to identify key characteristics of a large area (USDA 1996). The first step in this identification process was to determine the major concerns about the system emanating from social, economic, and ecological perspectives of the eight-state region. The assessment focused on terrestrial, aquatic, atmospheric, and social/cultural/economic conditions. Thus, the assessment was concerned with the condition of the natural resources as well as how people use the resources and their expectations. Because the SAA covers such a large area and such broad topics, a list of key terrestrial characteristics was developed for categories of forest health, wildlife and plant species, and important habitats. Aquatic characteristics include water quality, aquatic species, and habitats. The influences on ecological conditions of historical disturbances, land uses, and social and political forces were also considered, and both local environments and landscape perspectives were evaluated.

Once the important characteristics of a system are identified, the typical range of variation in those characteristics can be established within the focal and context areas and times. This information on the range of terrestrial, aquatic, atmospheric, and social/cultural/economic conditions provided the bulk of the five-volume Southern Appalachian Assessment (USDA 1996). The variability in these characteristics can be presented with regard to changes over time, environmental gradients in the area, or different levels of anthropogenic influences.

In their consideration of key characteristics, military natural resource managers have focused on endangered species and systematic inventories of vascular plant and wildlife. For example, the Army has instituted the Land Condition-Trend Analysis (LCTA) program as a standardized way to measure, analyze, and report data from inventory plots on plant communities, habitat, disturbances, impacts of military training, soil erosion potential, allowable uses, and restoration needs (Diersing et al. 1992). The purpose of that program was both to characterize the vegetation and to monitor change and detect trends in natural resources (Bern 1995). Sample plots were established in a stratified random manner using satellite imagery. Because the military testing and training typically result in intense, local, and broadly spaced impacts, the LCTA plots often do not capture the



spatial distribution of the effects. For example, at Yuma Proving Ground, Arizona, about 60 to 70% of the plots had no land use over the period 1991 to 1993 even though the actual land use was more extensive (Bern 1995). Therefore, the LCTA approach needs to be supplemented by a scheme designed to focus on discerning impacts and to integrate over broad spatial scales. Yet to relate the characteristics to the impacts, the stress also needs to be identified.

### *Step 3: Identify Key Stresses*

Stress to an ecological system is typically defined as any anthropogenic action that results in degradation (e.g., less biodiversity, reduced primary productivity, or lowered resilience to disturbances) (Odum et al. 1979; Barret and Rosenberg 1981; Odum 1985; Mageau et al. 1995). Stress can be classified into four categories: physical manipulations, changes in disturbance regimes, introduction of invasive species, and chemical changes [a slight revision of Rapport and Whitford's (1999) categories that use "stress" for anthropogenic activities]. Physical manipulations include human activities that can change soil conditions or construction of structures. Human activities may also cause fragmentation or eliminate critical habitats for some species.

Changes in disturbance intensity, frequency, duration, and extent can have major impacts on ecological systems (Dale et al. 1998). Disturbances are considered to be those events that are not typical of a system. For example, fires within a fire-moderated system, such as the lodgepole pine (*Pinus contorta*) forest of the western United States, would not be a disturbance to the system (even though individual organisms are impacted) (Fahey and Knight 1986). It is the absence of such fires that may cause a disturbance, for fires are an integral part of establishment and development of community structure of these forests. Thus, disturbances must be considered with regard to the life history of the major organisms in the community.

The introduction of invasive species is a major problem in many ecological systems. Often these introductions are nonnative species that do not have predators or competitors within the new system and thus become out of control. These introduced species can physically override the presence of other organisms and replace them quickly. There are numerous examples of such replacements (Westbrooks 1998). Occasionally invasive species may take over because of the elimination of some physical or biological constraints that may have been in the system in the past. *Lonicera maackii* (Rupr.) Herder (Amur honeysuckle), a large invasive shrub introduced into the United States in the late 19<sup>th</sup> century, has naturalized in at least 24 eastern states. It is abundant in habitats ranging from disturbed open sites to forest edges and interiors. *Lonicera maackii* negatively impacts native species, especially tree seedlings and forest herbs. Open, disturbed forests (e.g., Fort Campbell, Kentucky, where training can open forest canopies) are especially susceptible to colonization (e.g., Deering and VanKat 1998).

Chemical changes in the environment typically occur as a direct result of human activities. Point sources of toxins that result from spills or groundwater movements are a common cause of such a chemical change. Air pollution can also cause widespread and non-point source solution changes in systems.

Stress can be depicted as a gradient or a threshold such as intensity of impact, duration of event, or frequency of impact. Stresses are ultimately what most management



plans are for, both preventively and retrospectively. Often, changes in characteristics of a system result directly from one or more stresses. Typically, stresses interact and may exacerbate conditions for biotic survival or maintenance (Paine et al. 1998). Multiple stresses may be simultaneously analyzed or considered one at a time, depending on the goal of the analysis.

The stresses on military installations fit into the four categories of physical manipulations, changes in disturbance regimes, introduction of invasive species, and chemical changes. The training and testing typical of most installations creates a diversity of physical stresses ranging from soil erosion to vegetation removal. Alterations to fire frequency and intensity are the most common form of changing disturbance regimes. In some cases (such as Eglin Air Force Base on the Florida Panhandle), a prior landowner controlled fires, and the Department of Defense is now reinstituting a regular fire regime. The introduction of invasive species is a common problem on most installations. At Fort McCoy, Wisconsin, the leafy spurge (*Euphorbia esula*) threatens to encroach into oak savannas and outcompete the wild lupine. Kudzu (*Pueraria thunbergiana*) is present on most military installations in the Southeast where it literally overgrows anything in its path. Chemical changes on most installations occur as point sources in areas devoted to intense military activities (e.g., painting of aircraft). Usually, these sites are considered sacrifice areas in terms of conservation goals. However, chemical control of introduced species or along roadsides can also affect ecosystem management.

#### *Step 4: Determine How Stresses May Affect Key Characteristics of the Ecological System*

Once the process of selecting potential issues and identifying ecological characteristics and stresses within the context and focal systems is completed, the indicator selection process moves into the more specific stage of indicator selection. The process of developing and evaluating landscape-based ecological indicators is large and complicated, varies by region, and requires conceptual and causal links between stresses and the resulting ecological change (Brooks et al. 1998). Each concern that has been determined through the issue identification process needs to be analyzed in order to identify associated stresses, the cause of those stresses, the scope of those stresses on the management area, and the resulting changes in the characteristics of the management area.

Stresses are important to an ecological system in that they can disrupt composition, structure, or function. To the extent that these changes alter key characteristics of a system, the effect is significant. For example, insects or pathogens can increase tree mortality, reduce growth, and eventually change species composition and habitat patterns. Yet stresses that disrupt rare communities may be of the greatest concern to composition. For example, in the Southern Appalachians, 84% of the federally listed species occur in 31 rare communities and streamside habitats (USDA 1996), which means that management for endangered species can concentrate on select sites. However, there are considerable challenges to managing large tracts of land on the basis of a few endangered species.

Matrices that relate stresses to key ecological characteristics may be the best way to depict the effect that human activity may have on a system. For example, matrices containing the ways that military use can affect different types of vegetation at Fort



McCoy, Wisconsin have been developed (Dale et al. 2002b). The focus is on vegetation structure of the ground layer and the shrubs and trees because the wild lupine on which the larvae of the endangered Karner blue butterfly exclusively feeds occurs in the ground layer, and the shrub and tree layers provide the oak savanna system in which the lupine thrives. Such a matrix brings attention to those characteristics that are likely to change under current stresses and, thus, provides a way to identify indicators.

In much the same way that the spatial and temporal scales of the focal and context areas need to be defined, so too do the spatial and temporal scales of the individual stresses. As a result, stress effects may be limited to certain places or times. For example, ozone damage to sensitive trees may be greater at higher elevations where sufficient moisture is available from cloud cover to prevent stomata closure and allow more ozone to be absorbed. As a temporal example, some organisms are only susceptible to stress during their dispersal phase, while stresses at other times have little effect. For example, tank activity at Fort McCoy, Wisconsin actually enhances the presence of wild lupine upon which the endangered Karner blue butterfly oviposits (Smith et al. 2001). Yet, tank activity during the larvae stages can kill the insect.

#### *Step 5: Select Indicators*

The selected indicators should reflect the criteria (discussed earlier) and identify stress effects on key characteristics of the system. In general, these criteria call for indicators that are sensitive to the identified stressors in the system, sophisticated enough to capture the ecological system complexities, and responsive to identified stressors in such a way that they can be easily measured and monitored. Knowing how the stresses affect the key characteristics of the ecological system assists in the selection of indicators.

The selection of indicators is best made in a hierarchical manner. The selection process is initiated by considering the entire area of interest. For most military applications, this perspective would entail the installation as the focal site and the present as the focal time. However, the larger spatial and temporal context should also be considered. Thus, examination of the major physical gradients across the landscape or region should consider topography, soils, geology, land-use history, disturbance history, patterns of water (streams, lakes, and wetlands), and human use (roads, trails, buildings, and training and testing sites). Often the vegetation type, size, or density reflects the combination of these physical forces and serves as a useful indicator of their strength. For example, at Fort Stewart, Georgia, the amount of hardwood ingrowth into longleaf pine (*Pinus palustris*) stands indicates the time since the last growing-season fire. Thus, the pattern of vegetation types, such as hardwood ingrowth, or other land covers should be evaluated to see if it portrays features of the landscape that are indicative of stresses at the site and that may affect the ecological properties of the site. At Arnold Air Force Base in Tennessee, the high degree of forest fragmentation is indicative of past timber-harvesting practices and may portend effects on neotropical migrants (Robinson et al. 1995).

Ideally the suite of indicators should represent key information about structure, function, and composition. Yet the complexity of the relationship between structure, function, and composition only hints at the intricacy of the ecological system on which it is based. Often it is easier to measure structural features that can convey information about the composition or functioning of the system than to measure composition or



function. Sometimes measures from one scale can provide information relevant to another scale. For example, the size of the largest patch of a habitat often restricts the species or trophic levels of animals that are able to be supported based solely on their minimal territory size (Dale et al. 1994). Analysis of patch size for Henslow's sparrow at Fort Riley, Kansas indicates that the largest patch on the installation supports a declining population (the population's finite rate of increase is less than one) (Dale et al. 2000).

After the landscape is analyzed, the ecosystem and the species levels should be investigated. This process of considering characteristics of the system and potential indicators in a spatially hierarchical fashion needs to apply to each gradient of importance at the site. Placing the information on a spatial or temporal axis provides a means to check that information at all spatial scales. Alternatively, it is important to include indicators that encapsulate the diversity of responses over time (so that one is not just measuring immediate responses of the system). All major gradients are included in the analysis. We have focused on spatial and temporal scales, but it is also useful to consider the representativeness of indices across major physical gradients (soils, geology, land use, etc.).

#### *Step 6: Test Potential Indicators Against Criteria*

A crucial aspect for legitimizing the selection procedures for ecological indicators is the establishment of a scientifically sound method of monitoring system change. Each of the potential indicators needs to be tested to determine if it effectively measures the system characteristics of interest and meets the other criteria for indicators. This test should follow scientific procedures (e.g., theory and hypothesis development, hypothesis testing with control comparison, statistically significant results, etc.). The working hypotheses should reflect how specific indicators measure changes in key characteristics under stress. Experiments should be designed to compare measures of the indicators and key characteristics with and without stress events. For example, the condition of these indicators both before, during, and after documented stresses can then be compared with similar data collected in control sites. Based on the results of the tests for each potential indicator, the final set of ecological indicators can then be selected that is believed to be the most effective combination of indicators for monitoring the characteristics of interest to the management planners. The statistical analysis of such indicators is a basic aspect of most statistical text books.

#### *Step 7: Select Final Indicators and Apply Them to the Decision-Making Process*

The final ecological indicators are selected based on the test in Step 6. Then, management can implement monitoring of the suite of selected indicators. Long-term monitoring is an essential part of all environmental management programs, with adjustment of management activities based on indicator information and its relationship to overall management goals. The process of linking management to monitoring is part of adaptive management that views management actions as experiments and accumulates knowledge to achieve continual learning (Holling 1978; Walters 1986).

Often the application of measuring indicators or of adding refinements to measures can occur very quickly. This implementation aspect is especially rapid on Department of



Defense installations where the mentality is to act. For example, after we had used soil, geology, and slope to identify the sites at Fort McCoy, Wisconsin, that the wild lupine could occupy (Dale et al. 2000), the environmental site manager modified his monitoring program for wild lupine to focus only on areas that the analysis indicated could support the plant. This modification allowed the monitoring program to focus on those sites of greatest importance.

### Case Study

The objective of this case study is to identify indicators that signal ecological change in intensely and lightly used ecological systems at Fort Benning. Currently, military training, controlled fires (to improve habitat for the endangered red cockaded woodpecker), and timber thinning are common management practices on the installation. All of Fort Benning has experienced some anthropogenic changes either from past farming, logging, absence of burning, or military testing. Because the intent is that these indicators become a part of the ongoing monitoring system at the installation, the indicators should be feasible for the installation staff to measure and interpret. The focus is on Fort Benning, but the goal is to develop an approach to identify indicators that would be useful at several military installations. Because some of these effects may be long-term or may occur after a lag time, early indications of both current and future change need to be identified. The intent of this identification of indicators is to improve managers' ability to manage activities that are likely to be damaging and to prevent long-term, negative effects. Therefore, a suite of variables is needed to measure changes in ecological conditions. The suite that we are examining includes measures of terrestrial understory and overstory vegetation, soil microbial biomass and community composition, landscape patterns, and instream physiochemical and biotic water quality conditions. Because of the limited space in this publication, for further details we direct the reader to the project web site:

([http://www.esd.ornl.gov/programs/SERDP/research\\_projects.html#conservation](http://www.esd.ornl.gov/programs/SERDP/research_projects.html#conservation)).

The analyses of vegetation data collected from sites at Fort Benning with five discrete land-use histories showed high variability in species diversity and lack of distinctiveness of understory cover and led us to consider life form and plant families as indicators of military use (Dale et al. 2002a). Life form successfully distinguished between plots based on military use. For example, phanerophyte species (trees and shrubs) were the most frequent life form encountered in sites that experienced infantry foot traffic training. Analysis of soils collected from each transect revealed that depth of the A layer of soil was significantly higher in reference and infantry foot traffic training areas which may explain the life form distributions. In addition, the diversity of plant families and, in particular, the presence of grasses and composites were indicative of training and remediation history. These results are supported by prior analysis of life form distribution subsequent to other disturbances (Adams et al. 1987; McIntyre et al. 1995; Stohlgren et al. 1999) and demonstrate the ability of life form and plant families to distinguish between military uses in longleaf pine forests.

The soil microbial community of a longleaf pine ecosystem at Fort Benning also responds to military traffic (Peacock et al. 2001). Using the soil microbial biomass and community composition as ecological indicators, reproducible changes showed



increasing traffic decreases soil viable biomass, biomarkers for microeukaryotes and Gram-negative bacteria, while increasing the proportions of aerobic Gram-positive bacterial and actinomycete biomarkers. Our results indicate that as a soil is remediated it does not escalate through states of succession in the same way as it descends following military use. We propose to explore this hysteresis between disturbance and recovery process as a predictor of the resilience of the microbial community to repeated disturbance/recovery cycles.

The landscape metrics for Fort Benning were calculated and analyzed, and an assessment was made of the accuracy of the land cover estimates obtained from remote sensing as compared to *in situ* observations of land cover (Olsen et al. 2001). Metrics at the class and landscape level were compiled and analyzed to determine which were the best indicators of ecological change at Fort Benning. A set of metrics was selected, based upon change through time or ability to differentiate between land cover classes. We found the most useful metrics for depicting changes in land cover and distinguishing between land cover classes at Fort Benning were percent cover, total edge (with border), number of patches, mean patch area, patch area range, coefficient of variation of patch area, perimeter/area ratio, Euclidean nearest neighbor distance, and clumpiness. An accuracy assessment was performed of the 1999 land cover classification that was created using a July 1999 Landsat ETM image as compared to a 0.5-m digital color orthophoto of Fort Benning taken in 1999. The overall accuracy was found to be 85.6 for the 30-m resolution data (meaning that 85.6% of the test sites were correctly classified).

Landscape metrics indicate that the forest pattern (particularly that of pine) has declined greatly since 1827 (e.g., the area of pine forest declined from 78% to 34% of the current installation). Altered management practices in the 1990s may have resulted in changes to the landscape at Fort Benning. Several trends, such as an increase in non-forested and barren lands in riparian buffers were slowed or reversed in the last decade. Pine forest, on the other hand, appears to have been increasing in the last ten years. Improved monitoring techniques coupled with an aggressive management strategy for perpetuating pine forest at Fort Benning may have resulted in an increase in pine populations and a decrease in hardwood invasion. This management strategy includes harvesting timber and burning to establish and maintain viable pine communities. While it appears that the percentage of non-forest land has been slowly increasing, the number of non-forest patches has increased tremendously in the last decade. In other words, the non-forest land has become more fragmented over time. Consequently, the size of these patches has decreased significantly.

We are evaluating the efficacy of several stream chemistry and biology parameters as indicators of disturbance associated with military training and natural resource management activities at Fort Benning. This work is based on the idea that stream ecosystems are sensitive to disturbances within their catchments because many disturbances alter the patterns of runoff, drainage water chemistry, and inputs of biologically important materials to receiving streams. In addition, stream ecosystems are important components of the landscape and indicators of disturbance to stream biological communities and biogeochemical processes are an important part of any assessment of ecosystem health. Our research uses a disturbance gradient approach in which 1<sup>st</sup>- to 3<sup>rd</sup>-order streams draining catchments with strongly contrasting disturbance levels have been selected for study. These catchments are distinguished by percent bare ground for some



have little disturbance and others have widespread erosion caused by regular tank traffic. The inclusion of several reference streams in our study design provides data on the range of values for physicochemical and biological parameters expected for catchments showing minimal level of disturbance. Data from streams along the disturbance gradient are being compared to evaluate the suitability and sensitivity of specific disturbance indicators. The potential aquatic indicators at Fort Benning have been narrowed to:

- Suspended sediment concentrations (both baseflow and storms) and baseflow ( $\text{PO}_4$ , DOC) and stormflow ( $\text{NH}_4$ ,  $\text{NO}_3$ , and  $\text{PO}_4$ ) nutrient concentrations (indicator of erosion and biogeochemical status)
- Diurnal dissolved oxygen profiles (indicator of in-stream metabolism)
- Streambed organic matter content (indicator of food or habitat), and sediment movement dynamics (indicator of in-stream habitat stability or quality)
- Macroinvertebrate populations and communities, including EPT richness, Shannon diversity, biotic tolerance indices, and Bray-Curtis similarity of disturbed and reference streams (indicator of biological response)

For example, storm sediment concentration profiles show that streams in highly disturbed catchments had much higher rates of erosion and sediment transport than streams in less disturbed catchments.

The effects of historical land use / disturbance on stream macroinvertebrates are also being examined. Using remotely sensed imagery from 1974 and 1999, we used the GIS extension ATtILA to estimate areal percentage of 1) bare ground on slopes  $>3\%$ , 2) successional stage of vegetation (early-regeneration forested land) on slopes  $>3\%$ , and 3) road density ( $\text{km road}/\text{km}^2$  catchment) for each catchment. These three land use variables were then combined to derive a disturbance index (DI), which was used to rank and compare each catchment's historic and contemporary disturbance level. With these data we are examining the degree to which current measures of biotic water quality relate to historical vs. contemporary disturbance conditions. Preliminary analysis indicated that percent silt in the streambed was positively correlated with levels of historical (1974) land use among the catchments. Moreover, relative abundance of macroinvertebrate functional feeding groups also was related to historical land use. Disturbance also resulted in lower richness of EPT (i.e., number of taxa within the aquatic insect orders Ephemeroptera, Plecoptera, and Trichoptera) than in reference streams but similar total richness of invertebrate species. These data indicate 1) a legacy of environmental disturbance in Fort Benning catchments that spans at least 25 years, and 2) knowledge of historical land use conditions may be critical in interpreting contemporary water quality conditions.

## Conclusions

Ecological indicators offer a means to measure the effects of resource management. A key challenge is dealing with the complexity of ecological systems. Criteria and procedures for selecting indicators offer a way to deal with this complexity. The Department of Defense is developing ways to implement the use of ecological indicators for ecosystem monitoring and management. The next step is implementing indicators into resource-management practices.



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### **Section 3 Background for SEMP Project: *Indicators of Ecological Change***

#### **1. Background**

The U.S. military has funded many programs that measure various components of an ecosystem (e.g., Land Condition Trend Analysis (LCTA)). In addition, models have been developed to simulate ecosystem components related to physical terrain and erosion, wildlife habitat and populations, biodiversity, and vegetation dynamics (e.g., the Army Training and Testing Area Carrying Capacity (ATTACC) model and installation-specific population models of red cockaded woodpecker). However, recently a SERDP-sponsored workshop on management-scale ecosystem research identified the need for indices of the status, “health,” or sustainability of an ecosystem (Botkin et al. 1997). This research was designed to meet that need.

Ecosystem management can be defined as “a collaborative process that strives to reconcile the promotion of economic opportunities and livable communities with the conservation of ecological integrity and biodiversity” (The Keystone Center 1996). As Jack Ward Thomas, former Chief of the U.S. Forest Service, said, “The 25 million acres of military lands are a critical part of an ecosystem approach, and we look to all branches of the military to provide leadership and guidance in a cooperative spirit as we together face the many challenges of this new and exciting approach in these historically critical times” (page 29 of Leslie et al. 1996). Jack Ward Thomas also points out, however, that the implementation of ecosystem management is still being explored and will continue to be developed and defined over time. Applying ecosystem management approaches to military lands is critically important because of the unique resources on these lands and fact that conservation issues may jeopardize military missions if not appropriately managed. Many definitions of ecosystem management have been proposed (Christensen et al. 1996, Agee and Johnson 1998, American Forest and Paper Association 1994, Bureau of Land Management 1994, Fitzsimmons 1996, Goldstein 1992, Grumbine 1994, Haeuber 1996, Lackey 1996, Slocumbe 1993, Wood 1994), but the challenge for DoD is to develop an operational definition that is useful to military land managers. The development of indicators of ecological change is a necessary first feature to the application of these concepts of ecosystem management. Relating these indicators to ecosystem health is also a challenge (Costanza et al. 1992, Interagency Ecosystem Management Task Force 1996). Military installations provide appropriate locations where a working theory of ecosystem management can be devised. Therefore, our task was to develop the indicators of ecological change at Fort Benning with the intent that these indicators be applicable, particularly, across the Southeastern United States’ military installations and potentially throughout the nation's DoD installations.

2. Historical influences on candidate indicators are discussed in the following report.



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## **Historical Influences on Candidate indicators**

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A SERDP Ecosystem Management Project Report

### **Introduction**

This report considers how historical activities may be affecting the suite of candidate indicators of ecological change that are being explored at Fort Benning, Georgia by researchers at Oak Ridge National Laboratory under the SERDP Ecosystem Management Project (SEMP). Because the land in southwestern Georgia is largely forested, we concentrate on forestry practices. Also, land management includes effects on both terrestrial and aquatic resources. We recognize that land management practices are influenced not only by changes in ecological understanding but also by legislation and technology developments. Therefore, ecological, legislative, and technological changes relevant to land management and forestry are considered.

The practice of land management has been greatly altered in recent years by the combined effects of changes in legislation, increasing environmental sensibility of politically active citizens, and improved scientific understanding of ecosystems. These changes have occurred throughout the world but are most dramatically reflected in renewed forestry schemes in the temperate forests of the U.S. and Europe.

### **Attributes of Land that People Value**

Managing land as ecosystems requires an awareness of the diverse attributes of the land and how they relate to human values. The benefits from land are many. Land serves as watersheds that provide purified water for rivers, streams, and municipal use. Ecosystems provide important habitats for wildlife. Dry forests have widely spaced trees interspersed with grass and shrubs that serve as grazing lands. Land offers many recreational opportunities, including hiking, camping, hunting, and fishing. Some people value land areas for their beauty and the spiritual experience they provide. Trees also supply the raw materials for paper, building supplies, furniture, and



pharmaceutical supplies. Forests are a critical part of the global carbon cycle, for trees store large amounts of carbon in their tissue and, in mass, provide the largest storage site for terrestrial carbon (Post et al. 1990). When the carbon cycle is out of balance, it can instigate global changes in temperature and precipitation. Finally, the land on Department of Defense installations provides places for military testing and training.

Although an economic benefit relates to some of these attributes, considering only the monetary aspects of the land is unsatisfactory. Even in the 1940s, Leopold (1949) discussed the difficulty of putting a value on land other than in an economic sense. He argued that many of the benefits arising from the natural interconnectedness of the environment are not appreciated until they are lost or at risk. Indeed, a gradual and largely unnoticed loss of ecosystem components frequently occurs until a threshold is reached at which point these losses impact human values. Once a threshold is reached, people cry out for a return to a system that provides some of the benefits that are now in jeopardy.

Too often, however, no short-term impact of these gradual losses of ecosystem attributes is obvious. People can substitute one organism for another, or may synthesize a pharmaceutical product that formerly came from a forest plant or animal. But lost ecosystem properties, aesthetic qualities, or spiritual attributes may be more difficult to mimic. Moreover, ecosystems provide and protect water quality, water yield, and habitats that support other flora and fauna. For example, the fungus-induced demise of the chestnut trees in the early twentieth century resulted in the loss of not only timber but also seeds, which are an important food for squirrels and other forest fauna. In addition, it changed nutrient turnover from a system dominated by chestnut trees (which have high nutrient turnover rate of the leaves and slow nutrient turnover of woody tissue) to one just the opposite (the chestnut oaks that largely replaced the chestnut trees have slow nutrient turnover in leaves, and the wood decays rapidly). Because this replacement caused such a complex alteration in ecosystem properties, the full extent of the changes may never be known.

Management policies reflect the changing values that people have for ecosystems. Thus it is important to determine how the ways that humans value land have changed over time. Throughout history, land has provided a source of agricultural products, building materials and fuel and as habitat for game. As undisturbed land became scarce in Europe, commercial forestry arose as a means to assure a sufficient supply of wood and wildlife. One goal of forest management is to maintain sufficient quality, quantity, and diversity of wood products for commercial uses. Tree diversity is an important aspect of management because the qualities of individual tree species affect their use. For example, Port Orford cedar grown in southwestern Oregon and northwestern California was preferred for venetian blinds and ship masts because of its strength when cut in long sections. Oak is preferred for floors because of its hardness. Douglas fir and pine make excellent structural material for buildings because they grow rapidly yet are strong. At any one time, the



forest industry seeks to have sufficient wood of appropriate size, species, and quality available for each of its mills distributed across the country. About 130 years ago, wood became a source of pulp for paper. Since that time, many tree plantations have been established to meet the need for pulp. Recently, the value of maintaining species habitats and forest health (e.g., resistance from fire and insect outbreaks) has been reemphasized. Great strides have been made in recognizing the ecological values of forests and in communicating to public and private owners how to preserve these values (Swanson and Franklin 1992, National Research Council 1992, Salwasser 1990, Kessler et al. 1992, Franklin 1989). Yet there are still gaps. For example, the public does not do a good job of integrating the many attributes of forests, nor have ecosystem scientists adequately demonstrated the integration process (Naiman 1996). Also, because no single stand maintains all these values simultaneously, a coordinated effort between federal and state land holders, private forest companies, citizens, and citizen groups is necessary to maintain an appropriate balance of forest resources within a region.

Adaptive management recently arose as a way to preserve diverse values amid complex land ownerships. This approach monitors how environmental decisions affect forests by tracking such ecosystem attributes as trophic dynamics, succession, and nutrient cycling (Christensen et al. 1996). The management is adaptive in the sense that the lessons learned from past actions and their effects are used to guide future actions.

Monitoring land attributes provides a challenge in itself. The USDA Forest Service in coordination with the Environmental Protection Agency (EPA) has monitored attributes meant to reflect the diverse values of forests, including the ecological perspective [as measured by EPA's Environmental Monitoring and Assessment Program (EMAP)] (Table 1) (Lewis and Conkling 1994). Despite the fact that these efforts were meant to cover a comprehensive range of ecological attributes, animal conditions in forests were largely ignored. Even so, the attributes included are too numerous and technical for easy measurement or for the general public to grasp a basic understanding of them.

### **Candidate Ecological Indicators for Fort Benning, Georgia**

Our selection of ecological indicators of change focused on the longleaf pine habitat which is the dominant forest type at Fort Benning today. We selected candidate indicators of change in ecological integrity caused by anthropogenic disturbance, specifically military training activities within the Fort Benning military installation. The broad objective was to establish a suite of ecological indicators to represent and monitor the ecological integrity of the longleaf pine habitat within the military installation. Our perspective is that a suite of indicators is necessary to capture the full spatial, temporal, and ecological complexity that should be measured (Dale and Beyeler, in press). We, therefore, have proposed a candidate suite of indicators for longleaf pine forests at Fort



Benning, Georgia, that together characterize the spatial and temporal scales of interest as well as the diversity in soils and other environmental gradients (Figures 1 and 2). The suite to be examined includes measures of terrestrial biological integrity, stream chemistry and aquatic biological integrity, and soil microorganisms as a measure of below-ground integrity of the ecosystem. Understory vegetation is the element of this suite thought to represent ecological changes that may occur over a few years to decades within a forest stand and should reflect the differences in military training regime.

The key stresses at Fort Benning today are primarily due to human activities. Infantry and tank training are the major military actions on the base. Timber extraction for fiber occurs in some locations. Historic fire suppression was replaced in the 1970s with an active fire initiation program largely targeted at enhancing sites for longleaf pine. Natural pine bark beetle outbreaks occur sporadically (and are often triggered by drought and occur in stand with a high density of susceptible trees). The stresses at Fort Benning primarily affect the key characteristics of the ecological systems by either direct removal or reducing the quality of the ecological systems.

Historically, the land on which Fort Benning is located was intensely farmed by native Americans and settlers of European origin (Kane and Keeton 1998). The Native Americans depended on crops for much of their food. They hunted, gathered fruits and butts, and cultivated large agricultural fields dominated by corn.

A map of the vegetation in 1827 has been developed from historical land survey maps (Figure 3). Prior to lands being made available for public distribution in the United States, they were surveyed by the General Land Office (now the Bureau of Land Management) and the location and type of corner trees and witness trees were mapped. We are using the survey maps and field notes to create a digital GIS model of the forests from the early 19<sup>th</sup> century for the Fort Benning area. This 1827 map provides the baseline conditions for the installation.

After the lots were freely distributed via lottery, some people consolidated several contiguous lots and were able to develop prosperous plantations (Kane and Keeton 1998). Slaves often worked on the largest acerages and landowners accumulated large profits by selling cotton. However, cotton also severely depleted soil nutrients. Extensive erosion also resulted from tree clearing and was aggravated by plowing. Farmers seldom used crop rotation, contour plowing, fertilization or other beneficial techniques.

Some of the land was acquired by the government in 1919 to establish Fort Benning. At least one home was moved to adjacent property. But soon afterward (in 1921) the family planted kudzu to help alleviate problems with erosion and to provide shade for the house. Thus some of the current problems with exotic species are founded in activities of the early settlers.



## **The Influence of Changes in Ecological Understanding, Legislation and Technology on Land Management**

The ecological, legislative, and technological context in which lands are managed is illustrated in Figure 4. By considering changes in these arenas, one can gain a better understanding of the forces that influence forest use and the effects of those forces on forest ecosystems.

### ***Ecological Advances***

Great advances have been made this century in the ecological understanding of how ecosystems change and interact with their environment. The roots of this understanding go back to the introduction of plant geography between 1765 and 1812, which explored the relationship between the spatial location of a forest and environmental gradients. In 1840, Liebig presented what is now known as the *Law of the Minimum*, which states that the growth and distribution of a species is dependent on the factor most limiting in the environment. [This theory was later modified by Shelford (1913) and Good (1931, 1953) to acknowledge that limitations act in concert with one another]. The first ecological textbook was authored by Warming in 1895. Merriam's concept of life zones (1894, 1898) set the stage for considering environmental factors responsible for the elevational and latitudinal distribution of species. Shortly thereafter, work by Cowles (1911) and Clements (1916) led to an interpretation of succession as the gradual changes in species composition and site conditions that occur in the aftermath of a disturbance. Gleason's (1926) perceptions on succession were contrary to the individualistic notions of Clements and led Gleason to introduce the idea of plant associations, which formed the basis for the concept of the *ecosystem*, a term coined by Tansley (1939). The concept of environmental gradients and their effect on plant distribution was demonstrated by Whittaker (1956).

The International Biosphere Program (IBP) marked an intense focus on biome properties that stretched from 1960 to 1974 (e.g., Van Dobben and Lowe-Connell 1975, Levin 1976, Waring 1980, O'Neill et al. 1975). Experimental forests were established and monitored that demonstrated the intricate ecological relationships within an entire watershed (Bormann and Likens 1981). The large amount of data collected and analyzed during this period was complemented by a set of ecosystem models that were developed and exercised to assess the major components and interactions of ecological systems. The area of systems ecology arose from the understanding that resulted from these efforts. Forest-simulation models were developed that considered various factors affecting forest growth and development (Botkin et al. 1972). The IBP also stimulated research in evolutionary ecology and plant demography.

During the past two decades several new directions in ecological analysis have arisen. Landscape ecology, for example, focuses on the spatial relations of sites and considers the size, shape, and location of ecosystems (Foreman and Godron 1986, Turner 1989, Turner and Gardner



1991). This perspective forces ecologists to consider more than just natural systems. The large spatial context of landscape ecology requires an acknowledgment of the role that anthropogenic influences have had on shaping the landscape.

More recently, the concept of a sustainable biosphere (Lubchenco et al. 1991) has led ecologists to consider ways in which human actions can result in the long-term viability of an ecosystem and the planet. The more general need to protect the habitats of species that may become at risk led to the development of *ecosystem management* as a tool to consider the diversity of resources within a forest (FEMAT 1993, Christensen et al. 1996, Slocumbe 1993). The basic concept of this approach is to maintain the critical functional aspects of the terrestrial and riparian systems. These practices are now being adopted by the land managers and lead to a holistic view of land values and management practices. Today's resource managers are challenged to implement successful ecosystem management (Carpenter 1996).

### ***Legislative Changes***

At the same time that ecological understanding was advancing, changes in legislation pertinent to ecosystems was occurring. Federally owned land in the United States is primarily managed by the Department of Defense (DoD), Forest Service (FS), National Park Service (NPS), Department of Energy (DOE), Fish and Wildlife Service (FWS), and Bureau of Land Management (BLM). Because the FS contains the largest forested land area and the DoD is the focus of this research, we focus on those land managers.

The US armed forces have had a formal mandate for managing natural resources since 1823 when timber and forest products were essential for shipbuilding (Siehl 1991). During the World Wars, millions of hectares were acquired by the military to house and train troops and to test technology. After the war, timber production, fire control, agricultural leasing, and hunting programs were initiated at many installations. A 1989 DoD directive called for balance among competing interests, and a 1994 memorandum required the maintenance and improvement of sustainability and native biological diversity of ecosystems while supporting the DoD mission. Today, while most military lands serve primarily for training and testing, DoD is proactively seeking to conserve biological diversity (Leslie et al. 1996).

Steps to preserve and manage federally owned forests arose from the growing conservation movement. Although forest reserves were set aside in the United States as early as 1891, the establishment of the USDA Forest Service in 1905 marks the formal start of forest management and the establishment of national forests.

Gifford Pinchot brought the German traditions of forestry to the United States, where he was instrumental in establishing the USDA Forest Service and became the first chief. The multiple forest uses that Pinchot instilled into the Forest Service are still maintained and provide the



underlying justification for having wilderness, recreation, and timber-management areas in the Forest Service lands of today. Yet, in many cases, noncommercial tree species are not recognized as valuable ecological members of the forest community; nor has there been long-term promotion of the nonmonetary value of forests (e.g., building soils).

During the twentieth century, a steady stream of legislation provided the means to establish and manage federally owned forests. The Weeks Law of 1911 authorized purchase of forests that serve as watersheds for important rivers. In 1924, the Clarke-McNary Act expanded forest management to include fire protection and permitted the purchase of land for timber production. The Forest Pest Control Act of 1947 provided federal cooperation for states and private landowners to manage insect pests and diseases. The Multiple Use-Sustained Yield Act was passed in 1960 to require national forests to produce a sustained yield of timber while protecting other forest resources (reflecting some of Gifford Pinchot's goals for forest management). In 1964, the Wilderness Act was passed, which preserves wilderness areas within national forests where no road or buildings may be built and timber harvest is prohibited. Because of numerous controversies that arose over the use of clear-cutting, the Forest and Rangeland Renewable Resources Planning Act of 1974 established procedures for the review and management of forests. In 1976, the National Forest Management Act and the Federal Land Policy and Management Act focused attention on the need to manage riparian areas (Gregory 1997).

Two laws that are generally applicable have greatly influenced forest-management practices on federal lands. The National Environmental Protection Act (NEPA), passed in 1970, requires formal review before changes can be made on federal lands. The Endangered Species Act, passed in 1973, requires protection of threatened and endangered species. Legal actions to protect the red-cockaded woodpecker and the spotted owl have dictated forest-management practices in forests containing those species. Basically, in some locations the forest managers must retain the forest structure necessary for these rare and endangered birds. Although these legal actions only pertain to sites where rare species occur, they have implications for forest management in general. In the case of the forests in the U.S. Pacific Northwest, a process has emerged that bases the management plan on socioeconomic values as well as ecosystem attributes of the forest (FEMAT 1993). This process involves deliberation between the interested parties and consideration of the spatial locations of forests of particular interest for specific values (Shannon 1997). For example, some locations are critical for timber harvest because of their location relative to sawmills. Other forests may have the old trees and large enough area to be important habitat for spotted owl.

The same conservation movement that spawned protection and management of federally owned forest also promoted responsible management of state and private forest lands. Between 1890 and 1917, state forest agencies were established, forest research and educational institutions were put in place, and pest-management programs were organized (Irland 1982). Beginning in the



late 1940s, increased housing and business development have threatened some suburban forests. But the new conservation movement of the 1960s sparked an emphasis on open space. The Land and Water Conservation Fund and the beautification programs established under Johnson's administration brought a new interest in forest acquisition for recreation and conservation. Beginning with the Oregon Forest Practices Act of 1972, some states have implemented laws regulating forest practices on state and private lands that provide some riparian protection and a diversity of criteria for forest management that varies from state to state (Gregory 1997). The adoption of a state forest-protection law is currently under debate in other states (e.g., Tennessee).

Today there is a conflict between the rights of private landowners and federal and state laws, epitomized by the Wise Use movement. Some private landowners feel that their management should be entirely open-ended, yet federal and state regulations may restrict some actions (e.g., in the case of wetlands). This conflict is an arena in which ecological knowledge can contribute to development of a land-use ethic that would guide the standards used by landowners in protecting the diverse values of forests

International trade practices and regulations also affect forest practices within countries. For example, the concern for using wood that has been harvested from sustainable plantations rather than from old growth has led for a need to define and certify sustainable-forestry practices. Also, forests and biodiversity issues received much attention as a part of the 1992 U.N. Conference on the Environment and Development (UNCED), which resulted in a number of instruments, including the Conventions on Climate Change and Biological Diversity and the Statement of Forest Principles. These agreements all reflect concern for the sustainable use and management of biodiversity in forests and put international pressure on developing a means of determining the diversity of forests and estimating potential effects of specific management policies (Stork et al. 1997).

### ***Technological changes***

Technological advances pertaining to use of natural resources have influenced what laws could be put in place and which aspects of ecological systems could be analyzed. In the early 1900s trees were cut by hand; horses or mules removed the logs from the forest, and rivers often provided long-distance transport (Karamanski 1989). Today's logging operations rely on machines. Paper mills established by the 1920s greatly increased the demand for wood, particularly from small trees. Motorized trucks became common in the forests in the 1920s. The railway expansion of the 1920s also had direct and indirect effects on forests. Trains could be used to transport trees from the forest to the mill. Also, trains allowed food products to be shipped from the Midwest, allowing the less-profitable agricultural lands in the eastern United States to be turned back into forests. However, in the process of establishing the railway system many trees were harvested and used for



railway ties. All of these technological advances expanded the area cleared and increased the number of trees harvested from a site.

The chain saw may be the invention that most influenced the amount of wood taken from a site. The chain saw was invented in 1926 but did not become prevalent in the United States until the 1940s (Hall 1977). The chain saw not only allowed more wood to be cut in a set time interval but also permitted more trees to be cut within a stand. Thus, not only was timber more easily harvested, but more branches and cull trees were cut for firewood or other purposes. Thus chain saws result in a “cleaner” forest but one with less dead wood, which serves as animal habitat and is part of the nutrient-cycling and water-purification processes. Although there was great concern that the chain saw would replace jobs, more people are now employed in the timber industry than were at the turn of the century. Also, chain saws have made the work of logging much more efficient and safe.

The economic value of chain saws was made clear in a recent survey of farmers in Rondonia, Brazil (Dale and Pedlowski 1992). In 1991, none of the 89 farmers we surveyed owned a chain saw. Yet shortly thereafter, the eventually successful candidate for governor of the state gave away chain saws as part of his campaign. The effect on forests of the rapid introduction of chain saws into the region is just beginning to be observed. The major tool for observing these changes is satellite imagery.

Satellite imagery is now commonly used in the planning process and to monitor changes in forest resources over time. The development of the normalized-difference vegetation index (NDVI) that relates reflectance, as measured in the image, to forest conditions was the breakthrough that made remote imagery a useful tool (Tucker 1979). The use of global positioning systems (GPS) now allows accurate recording of a location on the ground and thus augments the value of remote imagery. Satellite imagery is particularly important for international forestry. As late as 1990 (Dale 1990), we still did not know how much of the Earth’s surface was covered by forests. Also, the extent of large-scale natural disturbances and anthropogenic clearing was not known. For example, a large fire burned millions of hectares of land in southeast Asia, but the area was not even known to be burning until it was identified in a satellite image.

Land-cover products based on data from the MODIS (Moderate-resolution Imaging Spectrometer) will be available in April 2001. The MODIS Land Discipline Group (MODLAND) is currently producing data products such as spectral vegetation indices, net primary productivity, and land cover (Cohen and Justice 1999). MODIS has finer spectral and temporal resolution as compared to Landsat data, but the higher spatial resolution of Landsat is still needed for close examination of land-cover changes (Zhan et al. 2000).

The use of helicopters and small planes in forests became a part of some forest operations in the 1970s. Aircraft are used to distribute seeds, control fires, perform controlled burns, spray herbicides, spread fertilizers, and occasionally to harvest trees from otherwise inaccessible sites.



Because of their high cost, helicopters and planes are not routinely used except for cost-effective applications of herbicides and fertilizers.

The development of computers also changed the logging industry. The control room in modern paper mills contains sophisticated computers to monitor and alert the operators of all aspects of the papermaking process. Computers are also used to decide how to cut a tree to maximize the value of the wood and minimize waste. Computers allow for the development and use of detailed simulation models of forest development. These models project situations in which sets of species grow and compete for resources. Geographic information systems (GIS) are the hardware and software combination that allows spatially explicit information to be integrated into a map of site conditions. The combination of simulation models with GIS technology allows the spatial arrangement of trees to be analyzed. The models have been used both for improving understanding of forest interactions and for planning harvest and selection regimes. As mentioned previously, this spatial integration of information is critical to the adaptive management policy now being espoused for forest management (FEMAT 1993).

Machines for whole-tree harvesting are routinely used in Finland and Sweden to harvest trees in such a fashion that the soil and roots are not disturbed, minimizing impacts on soil runoff and nutrient cycling. The trees can be cut and lifted with these machines so that damage to adjacent trees is negligible, retaining the economic value of those trees, while preserving the forest as habitat for animals. Sometimes whole-tree harvesters are used at Fort Benning as well.

Overall some technologies (e.g., chain saws) have resulted in cleaner forests which are less likely to support ecosystem characteristics such as habitat and water purification. Whole tree harvesters may be the best example of a technology that retains many of the ecosystem features of forests.

### **Integrating the Landmarks in Land Management**

Together, these ecological, legislative, and technological advances during the past century allow land owners to implement a diversity of management practices. But just because the means are available does not always imply that the practice will be used. For example, helicopter logging now allows trees to be removed from slopes more steep and inaccessible than was previously possible, but helicopters are generally not cost effective and are rarely used for harvesting. In addition to technological advances, awareness of some basic ecological concepts as well as legislative restrictions and social mores frame today's management practices. A review of the ecological concepts taught to foresters and their management implications clarifies how ecological knowledge has helped forest managers understand the diversity of forces that determine which species grow where, how forests develop, and how disturbances affect forests. These concepts were derived by examining forestry text books (e.g., the text by Oliver and Larsen 1996).



**Gradient Concept:** Environmental gradients are partially responsible for the distribution of species, ecosystems, and biomes. **Use:** Sometimes silvicultural practices take advantage of the ecological understanding of gradients and encourage planting or maintenance of species mixes and densities appropriate for a site and a region. Often however, these natural gradients are ignored, and monocultures of trees are planted over large areas.

**Limitation Concept:** Leibig (1840) introduced the concept that the growth and distribution of species depend on the most limiting environmental factor. This *Law of the Minimum* is now understood to imply that most factors provide both upper and lower limits and frequently factors act in concert. **Use:** Forest management for sustained yield depends upon this concept. For example, thinning regimes are put in place to minimize competition for scarce resources and thus to maximum tree volume for a given stand age.

**Disturbance:** Disturbances, such as fire, windthrows, and insect outbreaks, are a natural part of many forests and affect forest structure, composition, and function. **Use:** Forestry, more than any other area of resource extraction, manages for the long term (with rotation times ranging from 10 to 110 years). Thus, foresters must recognize the potential for a disturbance to occur within a management cycle. Practices are in place to either manage the disturbance (e.g., by fire breaks) or manage the system so that it is less susceptible to a disturbance (e.g., by reducing the amount of flammable litter or thinning trees so they are less likely to be attacked by beetles). To maintain disturbance-prone ecosystems, it is recognized that eight natural or anthropogenic disturbances are necessary. Thus, at Fort Benning, fires are regularly set to burn the understory of long leaf pine forests in order to retain their essential features necessary for the threatened and endangered red cockaded woodpecker.

**Succession:** Succession is the gradual replacement over time of one species by another that results in an increase in biomass in forests, a change in species-life-history characteristics, a change in light levels, a switch from high to low net primary production, and an alteration of the storage of nutrients from the soil to the standing biomass. **Use:** Knowledge of successional processes pervades forest management. Some practices create initial stages of succession by spreading seeds, planting seedlings, or leaving seed trees in place. The overall goal of silviculture is to artificially move stands through seral stages by such activities as thinning. Harvesting is scheduled to maximize profitability of timber yield by cutting trees prior to their peak in net primary production. Harvesting trees at a time when much of the nutrients are still being cycled within the soil and leaving branches onsite reduce the loss of nutrients from the site.



***Watershed Concept:*** Forests cleanse water and reduce runoff. Changes in the upland forests affect the riparian system. ***Use:*** Some management practices are designed to enhance water quality and quantity, reduce soil erosion, and maintain the riparian system. Trees are not cut on steep slopes or adjacent to rivers. Downed branches are left onsite, and the litter layer is disturbed as little as possible. Stream-bank corridors of trees are left in place, and logging roads are maintained by providing proper drainage that reduces the chances of landslides into streams. Military activities are restricted within streams as much as possible. For example, rip rap is put in place for most tank crossings, and turning within stream beds is prohibited.

***Structure Concept:*** Many ecological attributes can be linked to structural elements of the forest (e.g., size of live and dead trees and canopy configuration). Thus, many functional and compositional features of the forest can be preserved by maintaining structural features. ***Use:*** Ecosystem management is largely built upon maintaining or establishing structural features. This approach suggests ways that forest products can be harvested or military activities retained while allowing some structural features, such as large trees or downed trees, to be left in place.

### **Recent Trends in Ecosystem Science**

The major advances in ecosystem science in recent decades have been built upon these concepts. In the 1960s, forest management operated under the belief that actions to promote timber production are not necessarily good for other forest-related values. However, ecosystem science demonstrated that structural complexity is an important feature of natural forest ecosystems and that complexity is created naturally through disturbances. Because these disturbances occur on a scale larger than a stand, a landscape view was adopted. Traditional practices of clear-cutting even-aged stands do not recognize the importance of structure or disturbance in shaping the forest. Major questions for ecologists that accept these concepts are, therefore, how much structural complexity is needed and how can management activities (e.g., logging) be arranged in time and space to promote these features. Finally, it is important to know the relationships between forest stands and nonforest sites and the relative importance of those relationships.

This review shows that many forest practices are built upon ecological concepts and vice versa. Much of what ecologists know about forests has been learned by working with foresters. Even so, a gap remains between current ecological understanding and forest practices. This gap is evident in practice where, for example, forest management is frequently targeted at timber harvest in small parcels, instead of considering multiple uses of the entire landscape. Unfortunately, some recent textbooks also focus on management for timber at a stand level rather than promoting ecosystem-management practices (e.g., Oliver and Larsen 1996).



However, new forestry approaches are being developed to promote practices that will enhance ecosystem properties (Kohm and Franklin 1997). For example, long rotations and variable-retention harvesting will both maintain ecologically important structural features of forests while allowing harvest to continue (Franklin et al. 1997). Alternatively, previously unproductive agricultural lands can now grow fiber for energy with intensive short rotations (e.g., 10 years). Proposals for these management practices take many forms, but one basic ecological impact is a return to the size structure more typical of natural forests, consideration of long-term and broad-scale implications of forest management, and the perspective of the forest as an interacting system.

The public understanding of ecosystem processes is also deficient. The public typically associates ecology with the environmental movement and does not understand critical aspects of ecosystems, such as feedbacks, changes over time and space, or the concept of a system itself. For example, Naiman (1996) points out that the public does not understand how water affects land processes and vice versa.

### **Challenges Ahead for Managing Lands as Sustainable Ecosystems**

Many current land management practices have developed from ecological studies, but it is also clear that the land management practices are not based on the most recent ecosystem science. Rather, forestry is a success story for ecosystem science in the sense that there is an established process for moving ecological theory into practice on DoD installations and national forest as well as in the private forest industry. However, this process takes some time and is largely based on personal interchange and experiences. An example of this transfer is given by looking at the authors of the recent book on *Creating a Forestry for the 21<sup>st</sup> Century* (Komb and Franklin 1997). Many of those authors are from the research branch of the Forest Service. A gap exists between the practicing foresters and the research foresters, but because some members of these two groups are in the same organization and professional societies, it is not as wide as it is in other fields.

Finally, a wide difference exists in application of forest-protection laws. For example, in the United States, some state laws protect streams, whereas other states have no such protection. The challenge for ecosystems scientists is not just to publish their papers, but also to present their new ideas and understanding in a way that is useful and applicable for resource managers.

Products from research efforts need to be evaluated in the context of reaching the relevant audience. Social scientists can help ecologists identify the characteristics of the interested parties and what forest values are important to them. Scientific papers may elegantly lay out results, but these results may not be attainable by land managers. Too many times our products are immersed in jargon, complicated analysis, or theory, with the management implication being unclear. A recent analysis of tools used by environmental decision makers shows that the results of research



efforts need to be available to managers and not just described in papers (Dale and English, 1999). Thus the challenges ahead for scientists are to find ways to deliver research results to the land manager. Therefore, the SEMP project is designed not only to develop and test ecological indicators, but also to see that they are implemented by the DoD managers at Fort Benning. Furthermore, this development and application is meant to serve as a demonstration for other DoD installations.

Irland (1992) has summarized future trends in land use: Suburban sprawl will continue. The values that the public attributes to ecosystems will change, with many people recognizing multiple uses of forests and others appreciating only wood products. Demand for wood and forest products will increase with a corresponding rise in prices that will add to existing pressures to harvest forests. Speculative booms in rural land will diminish the size and increase the turnover of privately owned forest parcels. Less private land will be available for hunting, fishing, and hiking. Mechanization of the forest industry will reduce the number of jobs. At the same time the military needs for land on which to test and train will continue.

Up to now, for some small landowners forest management for timber in the United States has reflected the lack of wood shortage. Future trends will exert pressures to manage forest lands more efficiently. Currently, ineffective thinning, sloppy harvesting, and inappropriate clear-cutting led to loss of productivity. Of the 21% of U.S. land that is managed as commercial forest (195 million hectares, 480 million acres), only 8% was seeded or planted between 1950 and 1978 (Oliver and Larsen 1996, p.7). The rest was allowed to regenerate naturally. Rapid turnover in land ownership, land fragmentation, and land-use conversion result in loss of forests or erode the value of forests for wildlife habitat or as watersheds. Large landowners such as DoD and USDA, on the other hand, have been increasing productivity significantly through improved genetics, fertilization, planting, and more efficient harvesting.

Ecological science offers a long-term and broad-spatial perspective that is essential to sustainable management of forest resources. The long view instills a sense of responsibility for the forest resources. Forestry companies and federal agencies that own their lands, rather than lease them, tend to cut and replant the forest in such a way that promotes long-term benefits. When land management is performed for the long term, the ecological interactions are an integral aspect of the changes in the forest. Yet even in the short term, terrestrial and aquatic resources can be appropriately managed based on a knowledge of succession, effects of gradients, environmental limits, and the roles of disturbance and watershed properties. It is the responsibility of science to educate land managers about how these principles come into play in forest management, and it is the responsibility of the public to challenge land managers to develop an ethic that can preserve the ecosystem features of the forests. Ethical resource management built upon sound ecological principles is the key to appropriate forest stewardship. Having ecological indicators to monitor key



changes in ecological system is basic for such a resource management system.

### **Acknowledgments**

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Table 1. Example indicators, dimensions, and values for forest systems developed by EPA and the USDA Forest Service.

| Indicators | Dimensions | Values |
|------------|------------|--------|
|------------|------------|--------|



|                      |                      |                      |
|----------------------|----------------------|----------------------|
| Soil acidity         | Forest type          | Aesthetics           |
| Foliar chemistry     | Tree size            | Recreation           |
| Soil class           | Visible plant damage | Cultural importance  |
| Crown measures       | Fragmentation        | Religious importance |
| Leaf area            | Visibility           | Consumptive use      |
| Branch evaluations   | Recreation use       | Services of nature   |
| Plant diversity      | Extractive use       |                      |
| Ozone bioindicators  |                      |                      |
| Lichen diversity     |                      |                      |
| Visible damage       |                      |                      |
| Vegetation structure |                      |                      |

## List of Figures

Figure 4. Key events in land management. Acronyms are as follows: NEPA (National Environmental Protection Act ), NFMA (National Forest Management Act), FLPMA (Federal Land Policy and Management Act), NDVI (normalized-difference vegetation index), GIS (geographic information system), GPS (global positioning system), and MODIS (Moderate-resolution Imaging Spectrometer) (based upon Dale et al. 1998).

Figure 1. Spatial hierarchical overlap of a suite of ecological indicators for Fort Benning, Georgia.

Figure 2. Temporal hierarchical overlap of a suite of ecological indicators for Fort Benning, Georgia.

Figure 3. Land cover map for Fort Benning in 1827



Figure 1

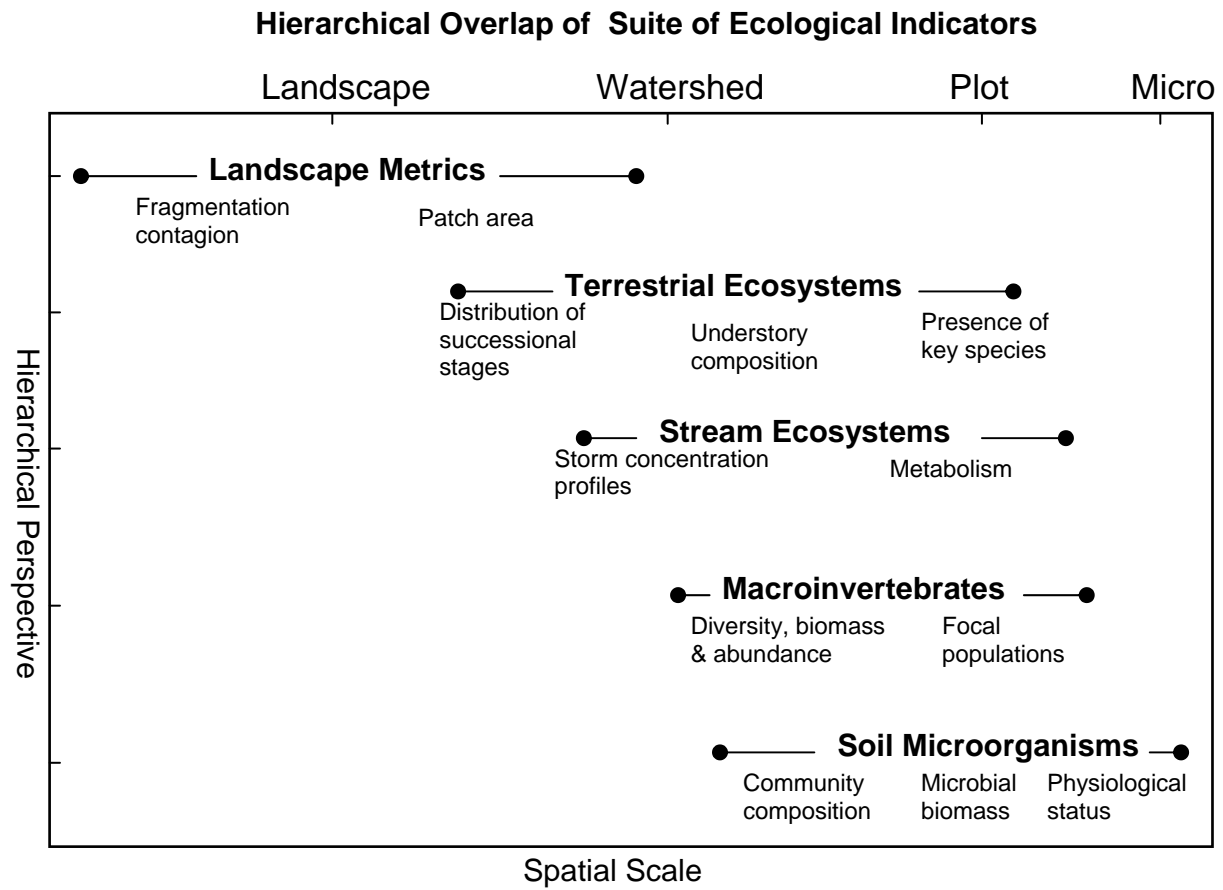




Figure 2

Hierarchical Overlap of Suite of Ecological Indicators Over Time

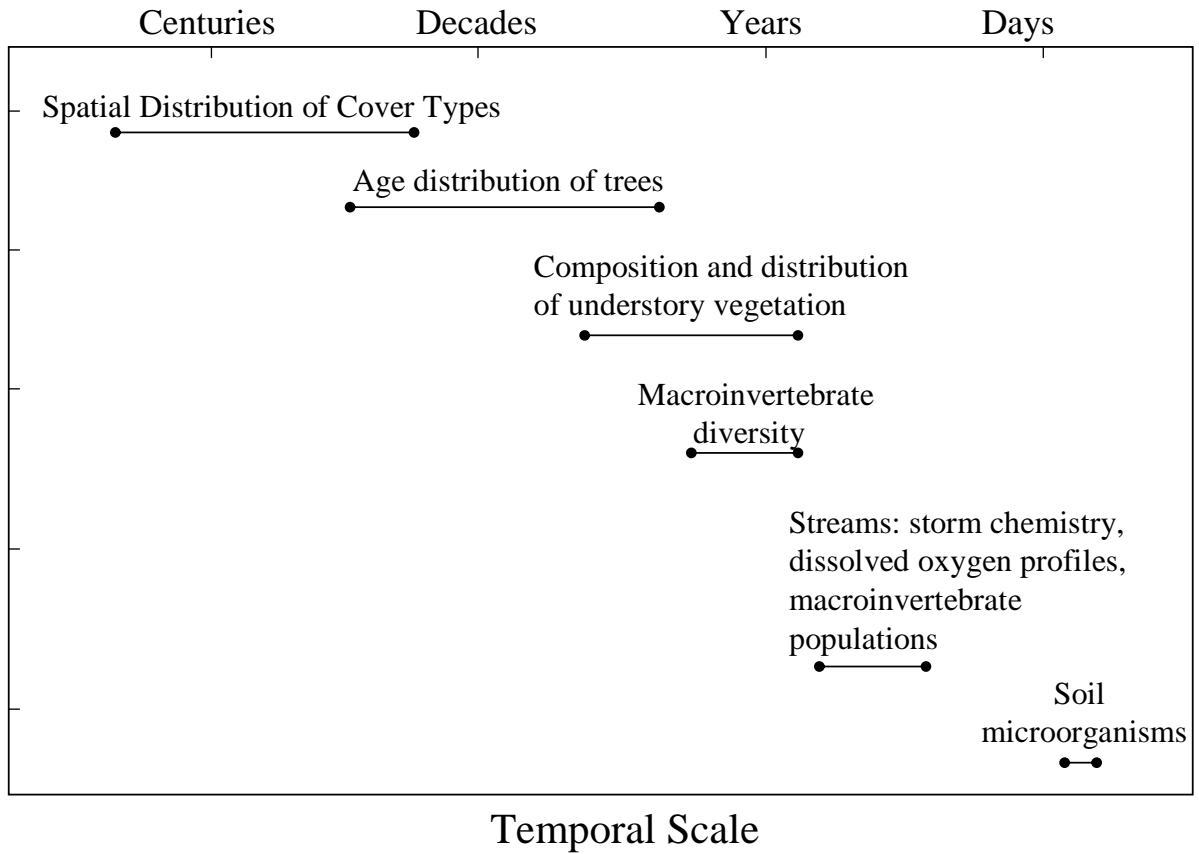




Figure 3.

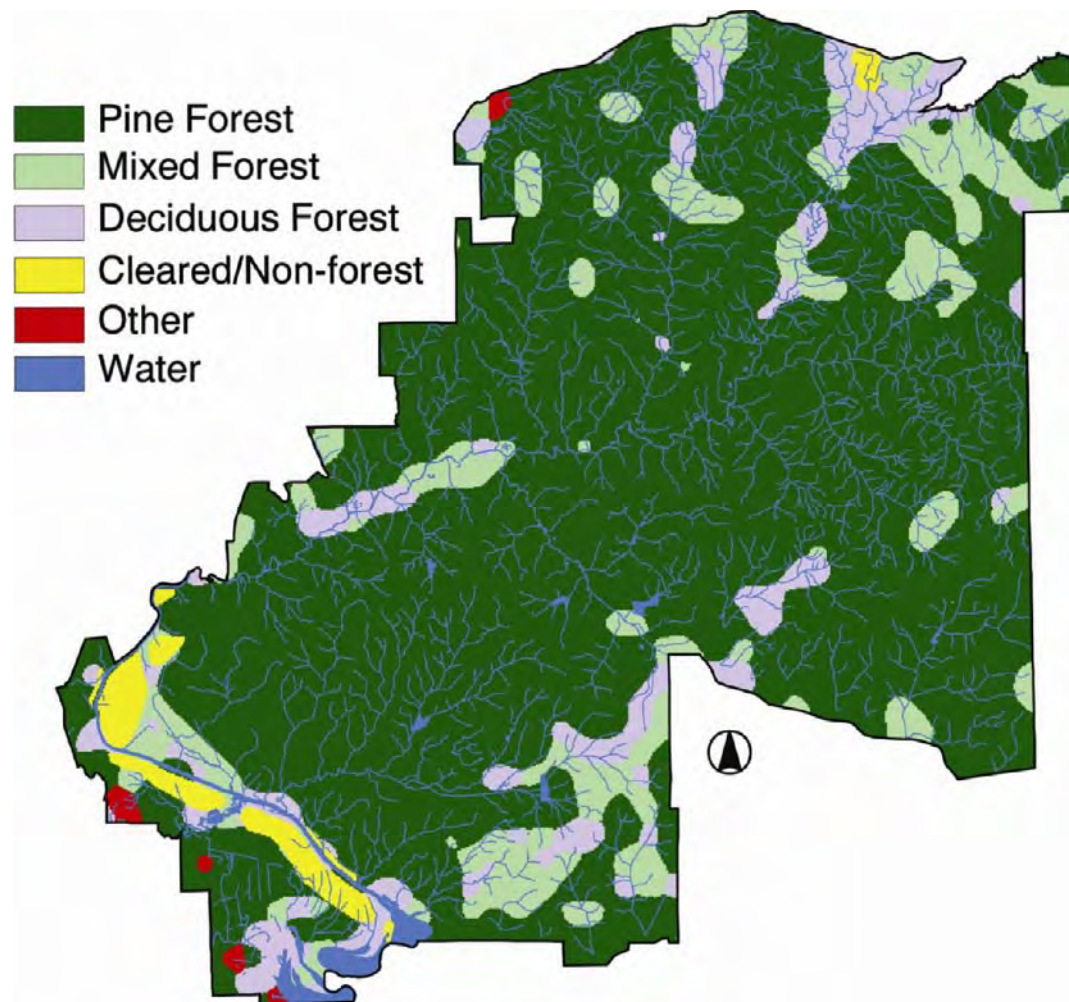




Figure 4

# Key Events in Land Management

| Year | ECOLOGICAL   | LEGISLATIVE  | TECHNOLOGICAL                               |
|------|--|--|---|
| 1900 | Lifezones<br>Succession  | Forest Service<br>Weeks Law  | Paper mills                                 |
| 1920 | Plant associations   | Clark-McNary Act   | Motorized trucks<br>Railway expansion       |
| 1940 | Ecosystem<br><br>Gradients   | Forest Pest Control Act  | Chain saws<br><br>Computers<br>Helicopters  |
| 1960 | IBP  | Multiple Use - Sustained Yield Act<br>Wilderness Act<br>NEPA and Endangered Species Act<br>Forest and Rangeland Renewable<br>Resources Planning Act  |   |
| 1980 | Landscape ecology<br>Sustainable biosphere<br>Ecosystem management | NFMA and FLPMA<br>DoD Directive calls for balance<br>DoD Memorandum on ecosystems<br>Conventions on Climate Change<br>Statement of Forest Principles | NDVI<br>GIS<br>GPS<br>Whole-tree harvesters |
| 2000 |  |  | MODIS                                       |



## **Section 4. Methods for SEMP Project: *Indicators of Ecological Change***

I. This research effort had five steps. These steps were to

- (a) Analyze historical trends in environmental changes to identify potential indicators;
- (b) Collect supplemental data relating to proposed indicators (building upon existing data already available at Fort Benning);
- (c) Perform experiments to examine how tank manipulations, troop movement, and other training or testing activities as well as restoration efforts at Fort Benning might affect these indicators;
- (d) Analyze the resulting set of indicators for the appropriateness, usefulness, and ease of taking the measure; and
- (e) Develop and implement a technology transfer plan.

Each of these steps is described in more detail below.

(a) **Analyze historical trends and environmental changes focusing on indicators and questions that deal with the role of these changes.** The goal of this analysis of historical data was to identify and map trends that have occurred over recent human history at Fort Benning and also to develop ways to measure these changes. The historical data on land use at Fort Benning was investigated for trends in metrics that relate to changes in ecological conditions over time, such as vegetation type and pattern. A second goal of this work was to characterize the disturbance history of different sites in order to evaluate relationships between the different indicators and level of disturbance which will be the focus of steps b and c. We used the information to establish sampling regimes and experiments in sites with relatively little disturbance and those that have been more intensively disturbed.

Long-term environmental changes at Fort Benning and other military institutions can be characterized with historical land survey maps. These maps were compiled as part of the federal and state public land distribution. As lands were opened up for public distribution in the United States, they were surveyed by the General Land Office (now the Bureau of Land Management). Among the states that formed from the thirteen colonies, lands were distributed at a state level. The states' process of distribution was similar to the federal system. Fort Benning is mostly bounded by land that was surveyed in the early nineteenth century by the state of Georgia.

The land distribution and survey system changed over time in Georgia. When the land bordered by Fort Benning was surveyed in 1827, land was divided into roughly equal lots within districts. Districts were to be roughly nine miles square and lots were to be 202 ½ acres square. Lots varied in size but were, on average, one half mile on one side. The lots were then issued at a lottery.

As part of the land distribution, the Surveyor General surveyed the land, noting the location of trees at each corner that marked the boundary of each lot. These trees were indicated on a map. The surveyor indicated the corner tree and four witness trees. The district maps are a sample of the forests during the early nineteenth century. In some instances, the surveyor measured the distance to each corner tree and circumference of tree. Using the "point quarter" method, a highly detailed estimation of the forest can



be reconstructed using this data. These maps and survey data provide an extremely valuable and unique method of characterizing long-term changes in the forests for a given region (see following report by Black et al. – Appendix A of this section).

A small section of Fort Benning is in Alabama which the federal General Land Office also surveyed. These surveys are slightly different than the Georgia survey but provide similar information. Surveyors in the Fort Benning area of Alabama only recorded corner trees so only species frequency can be determined.

We used the survey maps and field notes to create a digital GIS model of the forests from the early nineteenth century for the Fort Benning area. Although Native Americans had been living in the area for thousands of years, the model will be the best known representation of the forests in a pre-western agricultural environment. Thus, the 1827 map provided the baseline conditions for the installation.

An analysis of changes over space and time using GIS and statistical analysis of the changes in land cover over time was applied at Fort Benning. This approach provided a means to derive measures that are consistent across time and space. We focused on relationships between mapped features and the ecological systems that change significantly over time and space. These significant changes allowed us to examine conditions under which threshold of indicator values may be related to changes in ecosystem conditions. The metrics were examined more closely by focusing on the overall condition of terrestrial systems, aquatic systems that drain highly disturbed terrestrial areas, and on soil microorganisms (as described below).

**(b) Collection of data to supplement existing data and information already being obtained.** This analysis had three parts: examining terrestrial vegetation conditions, the biochemistry of land/water relationships and aquatic biological integrity, and shifts in soil organisms.

**(1) Changes in the terrestrial systems**

The components of ecological condition that were the focus of the terrestrial study were vegetation and soil microorganisms and landscape metrics. Using data already being collected by Fort Benning resource managers and information obtained as a part of this study, we examined the usefulness of the suggested indicators in providing information on sustainable ways to manage and use the DoD lands.

Indicators of fragmentation were examined to understand changes at the landscape level. Landscape metrics were selected that describe a diversity of land use types and how they may affect land management activities. These metrics cover a diversity of features about landscape patterns that can relate to how changes in fragmentation affect biological integrity of terrestrial systems. The estimate of landscape metrics used the 1827 land use map to provide baseline conditions. Because the landscape characteristics often change over time, their effects on the species or systems of concern may alter.

**(2) Changes in biological integrity of aquatic systems: land/water interactions and stream biological communities and processes**

This portion of the proposed work addressed several fundamental questions concerning aquatic indicators of disturbance to the biological integrity of streams: What are the chemical pulses in the streams that might indicate changes in the terrestrial



portions of the landscape? What are the indicators of changes in biological integrity of stream ecosystems resulting from disturbances in terrestrial ecosystems and changes in the amount and timing of materials transported from the land to water?

Many disturbances in terrestrial ecosystems result in changes in chemical outputs to streams draining these systems. The use of stream chemistry to identify changes in terrestrial ecosystems is central to the well-established watershed approach to ecosystem analysis. Disturbances in terrestrial ecosystems often produce increases in nutrient fluxes in streams. One of the most sensitive stream-chemistry indicators of disturbance appears to be the temporal pattern of nutrient concentrations during storms. Disturbances often result in sharp increases in nitrate and phosphate concentrations during storms, whereas nutrient concentrations in relatively undisturbed catchments tend to change very little or decline during storms. Concentrations of inorganic sediments also tend to increase sharply during storms in disturbed landscapes, in contrast to much smaller increases in minimally disturbed systems. As seen in some of our recent studies in streams in eastern Tennessee, changes in storm nutrient and sediment concentration profiles can occur at disturbance levels that are too small to result in appreciable changes in concentrations during non-storm periods (baseflow). Thus, storm nutrient and sediment profiles potentially are more sensitive disturbance indicators than baseflow nutrient concentrations. Further, because concentrations during high flows have a disproportionately large effect on total chemical output (which are the inputs to aquatic systems downstream), large increases in storm nutrient and sediment concentrations are of considerable concern even if baseflow concentrations are unchanged.

Disturbances to terrestrial ecosystems also often involve changes in hydrologic properties that result in a change in drainage-water flowpaths. Surface runoff is often increased as a result of reduction in the water infiltration rates of surface soils. Geochemical water-flowpath tracers are an important tool for identifying changes in water flowpath, particularly during stormflow. Changes in the concentrations of elements, such as calcium and silicon (which are often associated with deeper flowpaths) and sulfate and chloride (often higher in surface flowpaths), can be used to identify changes in catchment-water flowpath during storms (Mulholland 1993).

We evaluated storm chemistry profiles in streams as indicators of landscape disturbance. We selected small catchments with contrasting disturbance histories (highly disturbed, minimally disturbed) and compared storm chemistry patterns in each. This approach provided important information on the variability in storm chemistry responses to disturbance across a range of disturbance types and histories. The concentrations of nutrients (ammonium, nitrate, phosphate), potential flowpath tracers (calcium, silicon, chloride, sulfate), and other important water-quality constituents (suspended inorganic sediments, pH, dissolved organic carbon) in stream water were measured across the hydrograph for selected storms in different seasons.

Changes in terrestrial ecosystems can also have profound effects on the structure and function of stream ecosystems draining them. These effects may result from physical changes in riparian systems (e.g., vegetation removal, increased erosion) or from changes in material transport from land to water (e.g., nutrients, sediments, toxic substances). To assess changes to stream ecosystems resulting from terrestrial disturbances, we evaluated two types of indicators: stream ecosystem metabolism (i.e., rates of gross primary productivity, respiration at the whole-system level) and community-based indices of



biological integrity. Together, these metrics provide functional (metabolism) and organism-based (community) indicators of stream response to disturbance. As with our proposed evaluation of chemical indicators of disturbance, we compared streams with contrasting disturbance histories. Measurements were made seasonally to separate natural variation caused by seasonal biological cycles from disturbance effects.

Community-based indices of biological integrity integrate the composition, diversity, and functional organization of the community of organisms in the context of the regional climatic and physical/chemical template. These indices of biological integrity are potentially powerful indicators of disturbance effects because they quantify the integrated response of entire communities of organisms to changes in food resources as well as physical and chemical habitat characteristics. Several community-based indices of the biological integrity of stream ecosystems have been proposed. We evaluated several macroinvertebrate-based indices, for these indices have been shown to be sensitive indicators of disturbance to stream ecosystems.

The final aspect of the proposed work on land/water interactions and aquatic ecosystems involved technology transfer to DoD/Fort Benning personnel. Our intention was that by the end of year 5 of the research we would have identified those system level indicators that are most easily related to specific land uses and ecological implications. During the technological transfer phase of the project, we educated DoD personnel about the measurement and interpretation of the selected indicators.

### (3) Soil microorganisms as a measure of the below-ground aspect of ecological integrity

The soil biota is a useful measure of the functional integrity of terrestrial ecosystems. That is, soil microorganisms serve as indicators of the nutrient status and of the ongoing ecological processes that can potentially predict the future status of an ecological system. The objective of this task was to develop a methodology that measures shifts in microbial biomass, community composition, and physiological status (measured as signature lipid/DNA biomarkers) as a quick and easy indicator of changes in soil quality as a result of military activities and land management.

Soil quality ["the fitness of a specific kind of soil to function within its surroundings, support plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation"(USDA NRES 1996)] has the potential to be an effective index. Sustaining soil quality on military installations has many important benefits. Enhanced soil quality can help to reduce the on-site and off-site costs of soil erosion, improve nutrient use efficiencies, and ensure that the resource is sustained for future military use. It is also essential to maintain other resources that depend on the soil, such as water quality, air quality, vegetative productivity, and wildlife habitat. A technology that allows for the easy and cost-effective measurement and monitoring of soil quality could provide a valuable part of an overall adaptive land management tool with a goal of sustaining ecosystem health.

No single property is a sole indicator of soil quality, but the collection and analysis of many accepted soil quality indicators can be time-consuming and costly. Biological indicators of the changes in *in situ* microbial ecology, such as viable biomass, community composition, and physiological status, have been shown to correlate well to changes in other soil quality indicators (i.e., soil porosity, texture, organic matter,



phosphorus concentrations, nutrient cycling, and concentrations of elements). Because changes in these microbial indicators reflect changes in other geophysical properties of the environment, they may serve as a good overall indicators of soil quality. Therefore, a methodology that uses these biological indicators is a useful tool for land managers to cost-effectively determine changes in soil quality that may result from human use of the site.

We investigated the potential use of signature biomarker analysis for early detection of changes in soil quality as a result of military land use and land management activities. This project also employed spatial analysis and artificial intelligence technologies to simultaneously analyze multiple variables and develop a soil quality spatial model that can be used to predict soil quality and ecosystem status trends.

**(c) A field experiment was performed at Fort Benning to determine the appropriateness and usefulness of these proposed indicators as measures of ecological change.** The experiment was designed to examine how both impacts and restoration activities affect proposed indicators and the processes associated with them. The field experiment consisted of two major treatment types that will be applied at different levels. In both cases, data were collected prior to the application of the treatments, as well as subsequently. The indicator information collected relative to concerns dealing with vegetation, aquatic chemistry and biology, and soil microorganisms. The main treatment was tracked vehicle use. Although all of Fort Benning has some land use impacts, we selected an area that has had fairly light use historically and subject part of it to two intensities of tank use. Several lessons have been learned through this process about how to better implement such a field experiment (see Appendix B of this section).

**(d) Analysis of the resulting set of indicators considered their appropriateness, usefulness and ease of taking the measure.** Appropriateness includes an evaluation of whether the combination of metrics supplies information about unique aspects of biological integrity and threshold conditions. In the case of duplication of information, that measure can be selected which best meet the other criteria of usefulness and ease of measurement. The term "usefulness" can be interpreted from both an ecological and a land management perspective. Does the metric tell us something that a manager can respond to? Is it likely to change over the time that the site is being managed (e.g., a few seasons or years)? Is it ecologically meaningful? The ease of measurement criterion boils down to whether the DoD staff can obtain a statistically valid measure with relatively little cost and with confidence in the repeatability of the measure. It was important to field test this ease of measurement because the Fort Benning staff have the responsibility of continuing the monitoring once the research is completed.

**(e) It was critical to develop and implement a technology transfer plan of the indicators that result from this project or the effort will have been wasted.** The Fort Benning staff and The Nature Conservancy were key contributors to this part of the proposed work plan. Once the indicators were selected by the research team, a technology transfer plan was developed. The technology transfer information was



presented to the Fort Benning resource managers and The Nature Conservancy staff who are on site at Fort Benning. The report was then revised based on their comments.

## II. Selection of Research Sites for “Indicators of Ecological Change”

### (a) Understory and soil microbes as indicators of military use

This study was limited to maneuver training areas and, thus, does not include firing ranges, ordinance impact areas, or cantonment areas. Our goal was to develop valid and repeatable measures of impacts of training on understory vegetation and soil microbes in sites suitable for longleaf pine forests.

Study site locations were on land suitable for longleaf pine growth. Determination of potential site locations was achieved through a combination of existing forest stand information (Personal Communication, Bob Lairmore 1999, Fort Benning, GA) and county soil surveys of the United States Department of Agriculture Natural Resource Conservation Service. We overlaid an image of the United States Forest Service forest stand classification onto USDA NRCS soil maps for the area of land within the Fort Benning boundary. A final map was then created depicting locations of soils associated with longleaf pine within the installation boundary, and study sites were selected from those areas. Longleaf pine stands currently comprise approximately 5800 ha of the total area of Fort Benning. Soils favorable to the establishment and growth of longleaf pine make up approximately 65,900 ha (about 90% of the total area).

The study was designed based on a stratified sampling methodology. The sampling sites were blocked into five training intensity categories: reference, light, moderate, heavy, and remediation. Reference areas experience little to no training activities and are often in exclusion zones around firing ranges. Light training areas were limited to dismounted training and individual orienteering activities. Moderate training areas occur adjacent to tank training zones and are, thus, exposed to some tracked vehicle maneuvers, as well as limited vehicle and infantry traffic. Heavy training areas are used exclusively for wheeled and tracked vehicle training exercises. The classification of each site was primarily based on historical records of training activity; however, due to the variability of training intensity over space, final site selection was achieved through field reconnaissance and discussions with the Fort Benning natural resource personnel.

The remediation area is located in the uplands of the McKenna Drop Zone that was cleared in 1988 and subsequently rehabilitated. It is currently off limits to military training and testing. Revegetation efforts involved liming, fertilizing, and seeding with mixtures of grasses and legumes selected to increase vegetative cover and reduce runoff rates [e.g., giant reed (*Arundo donax*), Bermuda grass (*Cynodon dactylon*), little bluestem (*Adropogon scoparius*), Maidenhair (*Panicum hemitomon*), Pensacola Bahiagrass (*Paspalum notatum*), Alamo switchgrass (*Panicum virgatum*), weeping lovegrass (*Eragrostis curvula*), Lespedeza Sericea (*Lespedeza cuneta*, Var. *Sericea*) and Lespedeza Interstate (*Lespedeza cuneta*, Var. *Interstate*)].

Three transects were located in each of the reference, light, moderate, and heavy training classifications, and two transects were located in areas classified as remediation. Each transect was randomly placed within the fourteen study areas. The transects were established at a random distance and direction from a selected location.



**(b) Stream sites**

Streams were selected based on the following criteria

- (1) 1st or 2nd order in size and evidence that they were perennial
- (2) several streams in catchments of each of the dominant soil types (clay-dominated and sand-dominated soils)
- (3) encompassing a range in disturbance severity as determined from maps, discussions with Fort Benning staff, and observations in the field
- (4) reasonable access from road or trail

We quantified disturbance severity at the catchment scale for each stream using the land use/condition maps that we developed. We evaluated several metrics of disturbance severity (e.g., percentage of bare ground in catchment, number of intersections of bare ground with ephemeral and perennial streams, average slope of bare ground).



# Combining environmentally dependent and independent analyses of witness tree data in east-central Alabama

Bryan A. Black, H. Thomas Foster, and Marc D. Abrams

**Abstract:** We reconstructed pre-European settlement forest composition across 13 000 km<sup>2</sup> of east-central Alabama using 43 610 witness trees recorded in the original Public Land Surveys. First, we interpolated the witness tree data to estimate broad-scale vegetation patterns. Next, we conducted species-site analysis on landforms, an approach that was dependent on underlying environmental variables yet better resolved fine-scale vegetation patterns. East-central Alabama was dominated by three community types: oak-hickory across the Piedmont physiographic province and valleys of the Ridge and Valley province, pine-blackjack oak on the Coastal Plain province and ridges of the Ridge and Valley province, and white oak-mixed mesophytic in stream valleys and floodplains. Witness tree concentration (trees/km<sup>2</sup>) was highly uniform across much of the study area. However, there was an unusually low concentration of witness trees in the southwestern corner of the study area, and an unusually high concentration in stream valleys. Another irregularity was the inability of surveyors to distinguish black oak and red oak. Overall, the interpolations provided an unbiased, yet broad-scale estimate of forest composition, while the species-landform analysis greatly increased resolution of forest cover despite the subjectivity of defining environmental variables a priori.

**Résumé :** Nous avons reconstitué la composition de la forêt avant la colonisation par les Européens sur une superficie de 13 000 km<sup>2</sup> dans le Centre-Est de l'Alabama à l'aide de 43 610 arbres témoins présents dans les inventaires originaux des terres publiques. Dans un premier temps, nous avons extrapolé les données des arbres témoins pour estimer la répartition de la végétation à grande échelle. Par la suite, nous avons effectué une analyse des relations entre espèces et stations en fonction du relief; une approche qui est dépendante des variables environnementales sous-jacentes et qui permet de déterminer encore mieux la répartition de la végétation à petite échelle. Le Centre-Est de l'Alabama était dominé par trois types de communautés végétales : chêne-caryer dans la province physiographique du Piedmont et les vallées de la province de faîtes et vallées, pin-chêne noir dans la province de la plaine côtière et les sommets de la province de faîtes et vallées et chêne blanc-espèces mélangées adaptées à des conditions moyennes d'humidité dans les vallées des cours d'eau et les plaines inondables. La concentration des arbres témoins (arbres/km<sup>2</sup>) était très uniforme dans toute l'aire d'étude. Cependant, il y avait une concentration anormalement faible d'arbres témoins dans le coin sud-ouest de l'aire d'étude et une concentration anormalement élevée dans les vallées des cours d'eau. Une autre irrégularité venait de l'incapacité des arpenteurs à distinguer le chêne noir du chêne rouge. Dans l'ensemble, les extrapolations fournissent une estimation non biaisée mais à grande échelle de la composition de la forêt alors que l'analyse des liens entre l'espèce et le relief augmente grandement la résolution du couvert forestier malgré que la définition a priori des variables environnementales soit subjective.

[Traduit par la Rédaction]

## Introduction

Witness (bearing) trees recorded in surveyors' notes have been extensively used to characterize presettlement forests throughout the eastern and midwestern United States (Lutz 1930; Kenoyer 1930; Siccama 1971; Lorimer 1977; Grimm 1984; Cogbill 2000). They provide not only a momentary glimpse of forest composition, structure, and species-site

relationships but also reveal dynamic processes including historic patterns of disturbance and the nature and extent of Native American influences (Grimm 1984; Ruffner 1999; Black and Abrams 2001). Witness trees are often the only source of quantitative data on presettlement conditions where all original forests have been cleared, and even where original forests persist, post-European settlement land-uses may have altered them from their original character (Bourdo 1956). Although certain biases and inconsistencies may occur in the data, witness trees represent an important resource for investigating the ecology of presettlement forests and assessing the effects of pre- and post-European settlement land uses.

Two major approaches exist for describing forest composition from witness tree data: environmentally dependent analysis and environmentally independent analysis. In environmentally dependent analysis, species-site relationships are determined for a set of edaphic and topographic vari-

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ables such as soil texture, soil parent materials, slope, aspect, elevation, or landform (Whitney 1982; Abrams and Ruffner 1995). Community composition may then be determined by grouping species that share similar patterns of association with environmental variables. Although this method describes both synecological and autecological relationships in the presettlement forests, there are some potential shortcomings. One of the most problematic is that investigators are forced to choose environmental variables *a priori* and might not select those that are most significant. Also, testing these species–site relationships requires complete and accurate base maps of soils, topography, and hydrography, which may not yet be available for some areas. Another major complication is that environmentally dependent analysis assumes that underlying environmental variables are the sole regulators of forest composition. The abundance of a species or community is unrealistically assumed to be constant on all sites with the same set of environmental characteristics, ignoring the potential effects of natural or anthropogenic disturbance.

By contrast to environmentally dependent analysis, only spatial patterns in the witness tree data are used to reconstruct presettlement forest composition in environmentally independent analysis. The underlying landscape is not considered, eliminating the subjectivity of selecting appropriate environmental variables (White and Mladenoff 1994; Manies and Mladenoff 2000). In the earliest form of this approach, witness trees were transcribed onto modern base maps and vegetation cover was estimated by hand. However, now with the proliferation of geographic information systems (GIS), a number of interpolation techniques have become available to quantitatively and objectively convert points into continuous coverages (He et al. 2000). Then, if estimates of species–site relationships are desired, they can be determined by correlating interpolated results with the underlying environmental variables. Despite the unbiased nature of environmentally independent analysis, a weakness is that the scale of the resulting maps is very broad. For example, in western Mackinac County, Michigan, the optimal grain was estimated at 1 mi<sup>2</sup> (1 mi<sup>2</sup> = 2.590 km<sup>2</sup>; Delcourt and Delcourt 1996). A scale that size could smooth over important environmental variations, especially in regions that are more topographically diverse. Several distinct landforms with distinct vegetation types could occur within a square-mile block. Another potential drawback is that interpolations aren't as effective with minor species or communities. Landform-dependent analysis can provide more information in the case of small samples.

No matter whether environmentally dependent or independent analysis is used, irregularities in the witness tree data could decrease accuracy in the results. One of the most important problems is that witness trees represent an unintentional sampling, which was neither random nor impartial (Bourdo 1956; Black and Abrams 2001). Surveyors may have improperly identified some trees and may have even been biased toward certain species based on criteria of size, vigor, abundance, bark characteristics, economic value, or any other personal preference (Lutz 1930; Bourdo 1956; Loeb 1987). Furthermore, witness tree densities (number of trees per unit area) can vary with topography, as has been shown in colonial (pre-1785) metes and bounds surveys that occurred throughout New England and the mid-Atlantic

(Black and Abrams 2001). Tests for species and size biases in Public Land Surveys have largely failed to detect any significance. However, problems with inaccurate species identification frequently occur and tests on spatial variations in Public Land Surveys have not yet been conducted (Whitney 1994). If possible, the witness tree data should be checked for identification errors and spatial irregularities to ensure proper interpretation of the presettlement vegetation.

In this study, we describe presettlement forest composition across more than 13 000 km<sup>2</sup> of east-central Alabama using 43 610 witness trees recorded in the original Public Land Surveys. Few witness tree studies have been conducted in the southeastern United States (Delcourt and Delcourt 1977; Schwartz 1994; Cowell 1995), and those in Alabama have encompassed areas less than the size of a county (Dietz 1959; Jones and Patten 1966; Rankin and Davis 1971; Shankman and Wills 1995). Our study area spans three physiographic provinces and six physiographic sections that all exhibit substantial topographic and edaphic variation. To capture vegetation patterns across these diverse landscape features, we first apply environmentally independent interpolation techniques to provide an unbiased characterization of forest composition on a broad scale. We then perform environmentally dependent analysis on landforms, which vary on a much finer scale. This second step better quantifies species–site relationships, increases the resolution of forest composition, and investigates the distributions of minor species with sample sizes too small for interpolations. Finally, we describe variations in the concentration of witness trees across the landscape, some of which correspond to landforms and others of which appear to be the consequence of settlement patterns. Such spatial irregularities have not yet been quantified in Public Land Surveys, yet they could significantly affect the results of witness tree analysis.

## Materials and methods

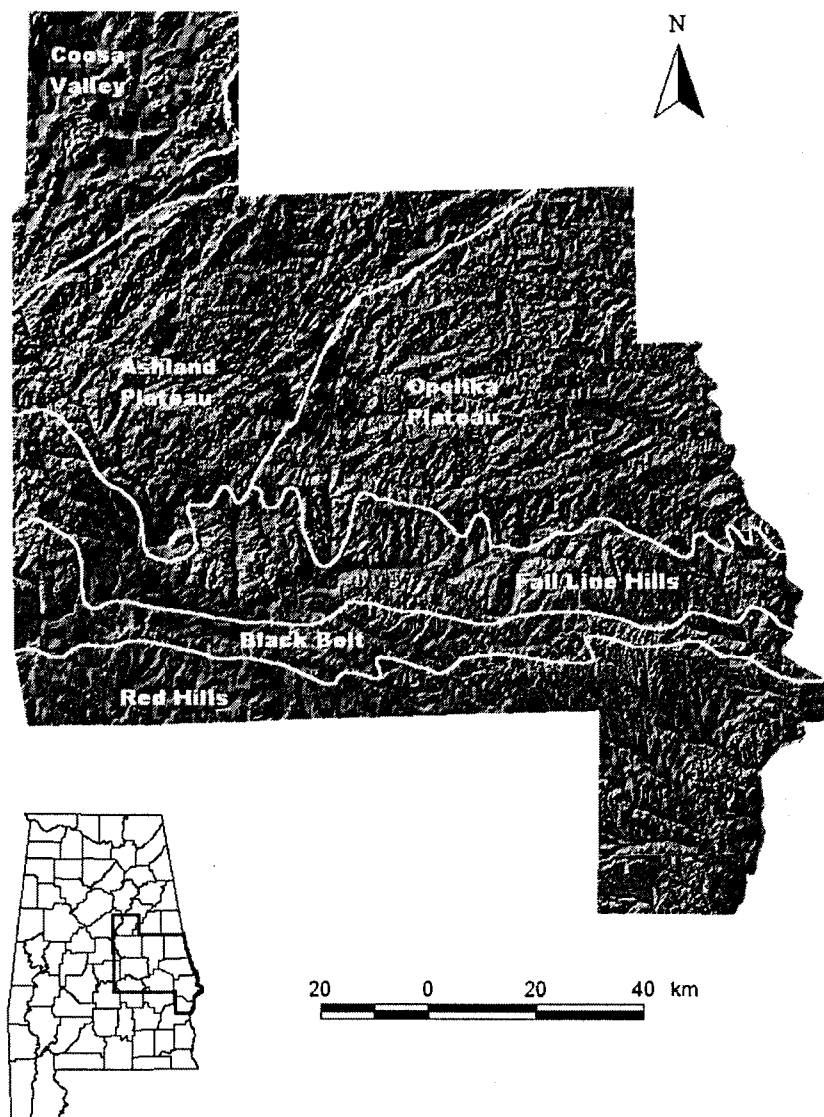
### Study area

The study region in east-central Alabama includes the Coosa Valley section of the Ridge and Valley physiographic province; the Ashland and Opelika plateaus of the Piedmont; and the Fall Line Hills, Black Belt (Black Prairie), and Red Hills sections of the Coastal Plain (Fig. 1; Fenneman 1938). The northernmost of these is the Coosa Valley section of the Ridge and Valley province. The long sandstone ridges that characterize this province are aligned on a northeast–southwest axis and interrupt gently rolling valley floors derived of limestone and shale. Elevations range from 100 m on the valley floors to 540 m on the ridges. To the south, stream valleys dissect the Ashland and Opelika plateaus of the Piedmont to form hilly uplands. The Ashland Plateau is somewhat more mountainous and xeric than the Opelika Plateau with several prominent ridges and a wider range in elevations (40–360 m on the Ashland Plateau and 60–270 m on the Opelika Plateau). The Ashland Plateau is derived from mica schists and slates, while schists and gneisses underlie the Opelika Plateau (Fenneman 1938).

South of the Opelika and Ashland plateaus are the three sections of the Coastal Plain, which are oriented longitudinally across the study area (Fig. 1). A transition from the Piedmont to the Coastal Plain, the south sloping Fall Line



**Fig. 1.** Topography and physiographic sections of east-central Alabama. The Coosa Valley is within the Ridge and Valley province; Ashland and Opelika plateaus are within the Piedmont; and the Fall Line Hills, Black Belt, and Red Hills are within the Coastal Plain.



Hills range in elevation from 40 to 200 m and are underlain by sandy and poorly consolidated sedimentary rock. In contrast, soils of the adjacent Black Belt section are formed of chalk residuum and are among the most productive in the southeastern Coastal Plain. Topographic relief is quite low with slightly elevated sandy patches occurring among more productive regions of high calcium carbonate and organic content (Fenneman 1938). More typical of the Coastal Plain is the sandy Red Hills Section, located in the southern part of the study area. Topography is diverse, ranging from hilly and dissected to broadly rolling. In addition, a number of wide, fertile stream valleys occur among the less fertile, sandy uplands. This section is named for its distinctively red-pigmented soils (Fenneman 1938).

The climate of Alabama is almost subtropical with temperatures ranging from a mean of 10°C in the winter to 32°C in the summer. Mean precipitation is about 137 cm. The mean length of the growing season is approximately 250 days with March 8 being the mean last day of frost and November 13 being the mean first day of frost (Burgess et al. 1960).

#### Witness tree surveys

The federal government conducted Public Land Surveys in Alabama during the 1830s–1850s, predating widespread European settlement. These surveyors divided land into square townships that were 6 miles (9.66 km) long on a side, and further divided each township into 36 square sections that were 1 mile (1.61 km) long on a side. Four markers were generally observed at each corner with two markers observed halfway between each corner at quarter-corners. Additional markers were usually identified, where surveyors intercepted major bodies of water. The nature of markers could be stones or posts, but by far the most common markers were trees, each of which were recorded by common name in surveyor notes. The result was a somewhat systematic, yet unintentional sampling of the forests at a density of approximately 24 trees per square mile.

#### Methods

Public Land Survey notes were obtained on microfilm from the United States Bureau of Land Management in



Washington, D.C. We digitized witness trees onto the Public Land Survey System digital line graphs (DLG) from the U.S. Geological Survey (USGS), using ArcView version 3.2a and ArcInfo version 8.02 (ESRI, Redlands, Calif.). Tree species common names were translated from Godfrey (1988), Schwartz (1994), and Cowell (1995) and are listed in Table 1.

We conducted environmentally independent analysis by first superimposing a 1-km<sup>2</sup> grid over the study area using ArcView version 3.2a, grid and graticule extension. We then tallied all witness trees within each cell with respect to species and entered the data into CANOCO version 4 (Microcomputer Power, Ithaca, N.Y.) for detrended correspondence analysis (DCA). Each of the 11 667 grid cells served as a plot. The DCA separated species into communities, and that information was used to generate a community map in which each 1-km<sup>2</sup> cell was classified according to its dominant community type.

As a second approach to environmentally independent analysis we interpolated the distribution of all major species, which we defined as those species with more than 900 individuals in the study area. For this method, we calculated the proportion of each major species within each 1-km<sup>2</sup> grid cell, providing a local estimate of species abundance while eliminating problems with the uneven concentration of witness trees across the landscape. Then, we converted the relative density grid theme to a point theme and performed inverse distance weighting interpolation to smooth the data using ArcInfo version 8.02 and ArcView version 3.2a. We felt that interpolating 1-km<sup>2</sup> grid cells would resolve the grain of the landscape in as much detail as possible while maintaining adequate sample sizes.

Next, we performed environmentally dependent analysis based on the six physiographic sections and landforms within each section. First, all trees were tallied with respect to physiographic section. Then, within each physiographic section, trees were tallied with respect to landforms. A total of four landforms were delineated on the Piedmont and Coastal Plain: north sideslope, south sideslope, hilltop-plateau, and stream valley – floodplain. The fifth landform identified in the Coosa Valley of the Ridge and Valley Province was valley floor. Landforms were identified by analyzing 1° digital elevation models (USGS EROS), the Alabama State Soil Geographic Data Base (STATSGO) digital soil layer (USDA 1991), and a hydrography layer.

The first step in delineating landforms was to define stream valleys and floodplains, which we identified as riparian zones and adjacent sites with soils derived from alluvial deposits. To generate this coverage, streams were buffered using ArcInfo version 8.02. Low-order streams (less than 5) were buffered to a width of 40 m, all other streams were buffered to 75 m, and major rivers were buffered to 100 m. Then, soils with alluvial parent materials were identified using the STATSGO digital soil layer (USDA NRCS 1991). Alluvial soils were combined with the buffered stream layer, and in some cases the floodplains were slightly extended in major river valleys where slope was less than 1%. Major floodplains only occurred in the Coastal Plain, particularly in the Fall Line Hills and Black Belt sections.

Sideslopes were all sites not included in the stream valley – floodplain coverage where slope was greater than 10%. All

other sites where slope was less than 10% and not already classified as stream valley – floodplain were designated hilltop – plateau. In the Coosa Valley section of the Ridge and Valley Province, we defined valley floors as any site not already included in the stream valley coverage where elevation was less than 180 m and slope was less than 10%. Level areas above 180 m were classified as hilltop-plateau. One-hundred-eighty metres was high enough to include almost all relatively level valley floors while excluding a large proportion of ridgetops and sideslopes.

Once all tallies were complete, we used contingency table analysis to quantify species relationships with environmental factors (Strahler 1978). Analysis was performed to determine species – physiographic section associations and then to determine species-landform associations within each of the physiographic sections. For each of these seven analyses, presence-absence tables were constructed for each species with one row for presence counts and another for absence counts. A column was constructed for each physiographic section or landform class. We used the *G* statistic to test for a significant species association with physiography and landform (Rohlf and Sokal 1995). Sample size for this statistic was deemed inadequate if at least two-thirds of the expected individuals had fewer than six individuals (Steel et al. 1997). For those species with a significant *G* statistic, standardized residuals were calculated for each landform or physiographic section according to the method of Haberman (1973). A positive residual indicates a positive association for the environmental factor, while a negative residual indicates a negative association (Strahler 1978).

#### Witness tree irregularities

We used density analysis to describe the homogeneity of witness tree concentration throughout the study area. The study area was divided into a 100-m grid, and all trees within 1000 m of each grid center were tallied. Tallies were converted to mean number of trees per square kilometre (ArcView version 3.2). In the seemingly irregular surveys in the southwestern corner of the study area, we also evaluated whether surveyors were biased toward marking trees on easily navigated landforms. To accomplish this, we first tallied landforms at each witness tree location. We then superimposed a 0.5-km<sup>2</sup> grid onto the study area and tallied landforms at each grid-line intersection. In the systematic grid tally, each landform should occur in roughly the same proportion as it does on the landscape. The potentially biased witness tree landforms were compared with the systematic tally using a  $\chi^2$  test. Finally, we tested the accuracy of surveyors' species names by superimposing townships and ranges onto the witness tree coverage. Surveyor biases or inaccuracies would be visible if a species abruptly increased or decreased in frequency across township or range lines (Cowell 1995).

#### Results

A total of 65 species was noted in the witness tree record, but overall, the region was dominated by pine, oak, and hickory (Table 1). The three dominant communities as separated by detrended correspondence analysis were pine – blackjack oak, oak-hickory, and white oak – mixed mesophytic (Fig. 2). Pine, blackjack oak, post oak, and hick-



Table 1. Common names and abundances of witness trees recorded in surveyors' notes.

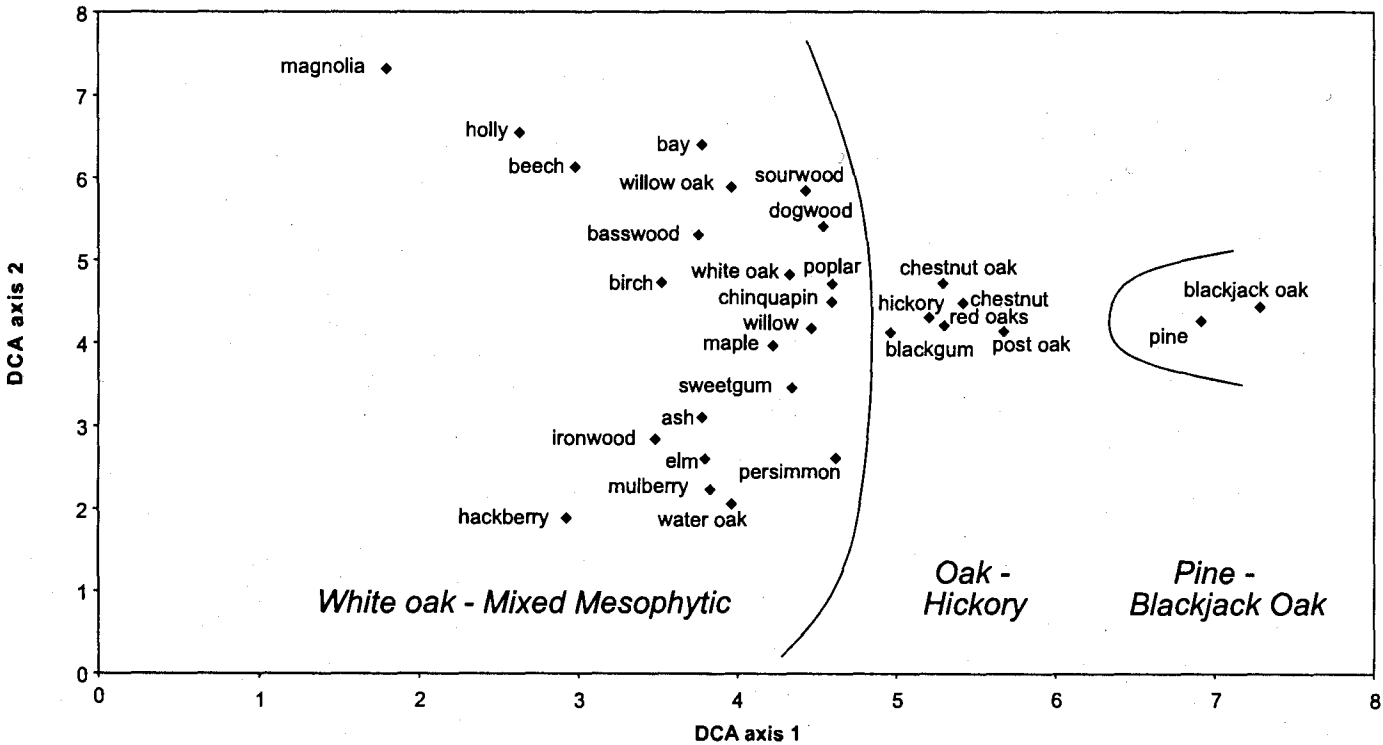
| Common name        | Scientific name(s)  | Count  | Relative density |
|--------------------|---|--------|------------------|
| <b>Gymnosperms</b> |   |        |                  |
| Cedar              | <i>Chamaecyparis thyoides</i> , <i>Juniperus silicola</i>   | 7      | 0.0              |
| Cypress            | <i>Taxodium ascendens</i> , <i>T. distichum</i>   | 7      | 0.0              |
| Pine total         | <i>Pinus echinata</i> , <i>P. palustris</i> , <i>P. elliotii</i> , <i>P. glabra</i> , <i>P. serotina</i> , <i>P. taeda</i> , <i>P. virginiana</i> | 19 184 | 44.0             |
| Spruce pine        | <i>Pinus glabra</i>   | 2      | 0.0              |
| <b>Angiosperms</b> |   |        |                  |
| Alder              | <i>Alnus serrulata</i>  | 9      | 0.0              |
| Apple tree         | <i>Malus</i>  | 3      | 0.0              |
| Ash                | <i>Fraxinus americana</i> , <i>F. pennsylvanica</i>   | 318    | 0.7              |
| Bass, linden, lynn | <i>Tilia americana</i>  | 87     | 0.2              |
| Bay total          | <i>Persea borbonia</i> , <i>P. palustris</i> , <i>Gordonia lasianthus</i> , <i>Magnolia virginiana</i> , <i>Magnolia grandiflora</i>              | 288    | 0.7              |
| Red bay            | <i>Persea borbonia</i> , <i>Persea palustris</i>  | 3      | 0.0              |
| Beech              | <i>Fagus grandifolia</i>  | 551    | 1.3              |
| Birch              | <i>Betula nigra</i>   | 39     | 0.1              |
| Black haw          | <i>Viburnum prunifolium</i>   | 2      | 0.0              |
| Cherry             | <i>Prunus americana</i> , <i>P. serotina</i> , <i>P. angustifolia</i> , <i>P. caroliniana</i> , <i>P. umbellata</i>                               | 27     | 0.1              |
| Chestnut           | <i>Castanea dentata</i>   | 1 013  | 2.3              |
| Chinquapin         | <i>Castanea pumila</i> , <i>C. ashei</i>  | 81     | 0.2              |
| Crabapple tree     | <i>Malus angustifolia</i>   | 3      | 0.0              |
| Cucumber           | <i>Magnolia acuminata</i> , <i>M. cordata</i> , <i>M. pyramidata</i> , <i>M. ashei</i>  | 20     | 0.0              |
| Dogwood            | <i>Cornus florida</i> , <i>C. alternifolia</i>  | 436    | 1.0              |
| Elder              | <i>Sambucus</i> spp.  | 3      | 0.0              |
| Elm total          | <i>Ulmus alata</i> , <i>U. americana</i> , <i>U. rubra</i>  | 183    | 0.4              |
| Red elm            | <i>Ulmus rubra</i>  | 16     | 0.0              |
| Blackgum           | <i>Nyssa sylvatica</i> , <i>N. aquatica</i> , <i>N. biflora</i>   | 738    | 1.7              |
| Hackberry          | <i>Celtis laevigata</i> , <i>Celtis occidentalis</i>  | 42     | 0.1              |
| Haw total          | <i>Crataegus</i> spp.   | 7      | 0.0              |
| Red haw            | <i>Crataegus spathulata</i>   | 1      | 0.0              |
| Hazel tree         | <i>Hamamelis virginiana</i>   | 2      | 0.0              |
| Hickory            | <i>Carya aquatica</i> , <i>C. cordiformis</i> , <i>C. glabra</i> , <i>C. ovalis</i> , <i>C. tomentosa</i>   | 3 826  | 8.8              |
| Holly              | <i>Ilex opaca</i> , <i>I. vomitoria</i>   | 239    | 0.5              |
| Honey locust       | <i>Gleditsia triacanthos</i>  | 2      | 0.0              |
| Huckleberry        | <i>Vaccinium arboreum</i>   | 10     | 0.0              |
| Ironwood           | <i>Carpinus caroliniana</i>   | 202    | 0.5              |
| Magnolia           | <i>Magnolia grandiflora</i>   | 37     | 0.1              |
| Maple total        | <i>Acer saccharum</i> , <i>A. rubrum</i> , <i>A. saccharinum</i>  | 447    | 1.0              |
| Box elder          | <i>Acer negundo</i>   | 15     | 0.0              |
| Sugar tree         | <i>Acer saccharum</i>   | 47     | 0.1              |
| Mulberry           | <i>Morus alba</i> , <i>M. rubra</i>   | 65     | 0.1              |
| Myrtle             | <i>Myrica cerifera</i> , <i>M. inodora</i> , <i>Ilex myrtifolia</i> , <i>Quercus myrtifolia</i>   | 1      | 0.0              |
| Oak total          | <i>Quercus</i> spp.   | 14 289 | 32.8             |
| Blackjack oak      | <i>Quercus marilandica</i> , <i>Q. laevis</i>   | 1 723  | 4.0              |
| Black oak          | <i>Quercus velutina</i> , <i>Q. falcata</i> , <i>Q. rubra</i> , <i>Q. shumardii</i>   | 2 541  | 5.8              |
| Post oak           | <i>Quercus stellata</i>   | 5 065  | 11.6             |
| Red oak            | <i>Quercus falcata</i> , <i>Q. velutina</i> , <i>Q. rubra</i> , <i>Q. shumardii</i>   | 2 943  | 6.7              |
| Spanish oak        | <i>Quercus falcata</i>  | 280    | 0.6              |
| Water oak          | <i>Quercus nigra</i>  | 227    | 0.5              |
| White oak          | <i>Quercus alba</i>   | 1 262  | 2.9              |
| Willow oak         | <i>Quercus phellos</i>  | 130    | 0.3              |
| Peach              | <i>Prunus persica</i>   | 21     | 0.0              |
| Pecan              | <i>Carya illinoensis</i>  | 3      | 0.0              |
| Persimmon          | <i>Diospyros virginiana</i>   | 128    | 0.3              |
| Plum               | <i>Prunus angustifolia</i> , <i>P. americana</i>  | 10     | 0.0              |
| Poplar             | <i>Populus deltoides</i> , <i>P. heterophylla</i> , <i>Liriodendron tulipifera</i>  | 313    | 0.7              |
| Red bud            | <i>Cercis canadensis</i>  | 13     | 0.0              |



Table 1 (concluded).

| Common name | Scientific name(s)   | Count  | Relative density |
|-------------|--|--------|------------------|
| Sassafras   | <i>Sassafras albidum</i>   | 211    | 0.5              |
| Sourwood    | <i>Oxydendrum arboreum</i>                                       | 213    | 0.5              |
| Sumac       | <i>Rhus copallina</i>  | 3      | 0.0              |
| Sweetgum    | <i>Liquidambar styraciflua</i>                                   | 461    | 1.1              |
| 7Sycamore   | <i>Platanus occidentalis</i>                                     | 23     | 0.1              |
| Wahoo       | <i>Euonymus atropurpureus</i>                                    | 15     | 0.0              |
| Walnut      | <i>Juglans nigra</i> , <i>J. cinerea</i>                         | 15     | 0.0              |
| Wild orange | <i>Citrus</i>  | 1      | 0.0              |
| Willow      | <i>Salix caroliniana</i> , <i>S. nigra</i> , <i>S. floridana</i> | 12     | 0.0              |
| Total count |  | 43 610 |                  |

Fig. 2. Detrended correspondence analysis of witness tree data using 11 667 plots (square-kilometre blocks) and 51 species. The 30 most abundant species are shown, and three communities are separated: white oak – mixed mesophytic, oak–hickory, and pine – blackjack oak. Axis 1 explains 3.6% of the species variance, while axis 2 explains 2.7%.



ory dominated the pine – blackjack oak community, while post oak, hickory, the red oaks (red oak, black oak, and southern red oak), and pine were the most abundant species in the diverse white oak – mixed mesophytic community. In general, pine – blackjack oak occupied the roughest terrain, such as the ridges of the Ridge and Valley and the most highly dissected regions of the Fall Line Hills and Red Hills (Figs. 1 and 3). The oak–hickory community dominated across the Piedmont, on the valley floors of the Ridge and Valley, and the level uplands of the Coastal Plain. The white oak – mixed mesophytic community occurred on stream valleys and floodplains throughout the entire study area (Fig. 3). Thus, axis 1 of the DCA can be interpreted as a moisture gradient from the moist, pro-

tected stream valleys to the gently rolling Piedmont and then to the roughest terrain of the sandy Coastal Plain (Fig. 2). Contingency-table analysis indicated both species of the pine and blackjack oak community were strongly associated with the Fall Line Hills, and to a lesser extent, with the Ridge and Valley. However, physiographic associations within the oak–hickory community were much more heterogeneous (Fig. 4). Post oak, the red oaks, hickory, and chestnut were positively associated with the Piedmont; however, the red oaks and hickory attained their peak abundances on the Opelika Plateau, while chestnut was associated with the Ashland Plateau. Chestnut oak occurred infrequently on the Piedmont but was a component of the oak–hickory forests of the Coosa Valley ridges (Fig. 4). By contrast, post oak was most strongly associated with the Piedmont and was only



**Table 2.** Relative densities of witness tree species in the presettlement forest communities of east-central Alabama.

| Species       | Pine –<br>blackjack<br>oak | Oak–hickory | White oak –<br>mixed<br>mesophytic |
|---------------|----------------------------|-------------|------------------------------------|
| Pine (total)  | 79.2                       | 12.5        | 8.8                                |
| Oak (total)   | 15.0                       | 56.8        | 28.2                               |
| Post oak      | 3.5                        | 24.9        | 2.8                                |
| Red oak       | 1.9                        | 14.2        | 2.9                                |
| Black oak     | 1.4                        | 13.0        | 2.2                                |
| Spanish oak   | 0.2                        | 1.3         | 0.4                                |
| Blackjack oak | 7.2                        | 1.1         | 0.5                                |
| White oak     | 0.6                        | 1.7         | 12.5                               |
| Willow oak    | 0.0                        | 0.1         | 2.0                                |
| Water oak     | 0.1                        | 0.2         | 3.3                                |
| Hickory       | 2.1                        | 19.5        | 3.6                                |
| Chestnut      | 0.6                        | 4.5         | 0.9                                |
| Blackgum      | 0.8                        | 2.6         | 1.2                                |
| Ash           | 0.1                        | 0.2         | 4.0                                |
| Basswood      | 0.0                        | 0.1         | 1.3                                |
| Bay           | 0.2                        | 0.1         | 4.1                                |
| Beech         | 0.2                        | 0.2         | 9.1                                |
| Birch         | 0.0                        | 0.1         | 0.6                                |
| Chinquapin    | 0.0                        | 0.1         | 0.8                                |
| Dogwood       | 0.3                        | 0.6         | 3.9                                |
| Elm total     | 0.0                        | 0.1         | 2.5                                |
| Hackberry     | 0.0                        | 0.0         | 0.7                                |
| Holly         | 0.1                        | 0.1         | 3.9                                |
| Ironwood      | 0.1                        | 0.1         | 3.3                                |
| Magnolia      | 0.0                        | 0.0         | 0.7                                |
| Maple (total) | 0.2                        | 0.4         | 5.3                                |
| Mulberry      | 0.0                        | 0.1         | 0.9                                |
| Persimmon     | 0.1                        | 0.2         | 1.2                                |
| Poplar        | 0.2                        | 0.5         | 2.6                                |
| Sassafras     | 0.1                        | 0.4         | 2.1                                |
| Sourwood      | 0.1                        | 0.2         | 2.2                                |
| Sweetgum      | 0.3                        | 0.4         | 5.1                                |
| Tree count    | 12 982                     | 4277        | 17 849                             |

weakly associated with the Ridge and Valley (Fig. 4). In the white oak – mixed mesophytic community, most species were associated with the Black Belt and Red Hills of the Coastal Plain where stream valleys and broad floodplains were most abundant (Figs. 1 and 4). Yet some species such as bay, beech, holly, and magnolia were more strongly associated with the Black Belt, while white oak, water oak, basswood, and elm were more strongly associated with the Red Hills (Fig. 4). This difference in physiographic association appears to explain a large part of axis 2 in the DCA (Fig. 2).

Patterns of standardized residuals summarize species associations with physiographic sections, but the actual distribution of each species is much more complex. Interpolations of major species illustrate how concentrations of species varied throughout physiographic sections in response to landforms and topographic roughness. Overall, the red oaks, post oak, and hickory occurred on gently rolling regions within each physiographic section, while pine and blackjack oak occupied the roughest terrain (Figs. 1 and 5). Indeed, pine attained its greatest concentrations on the most highly dis-

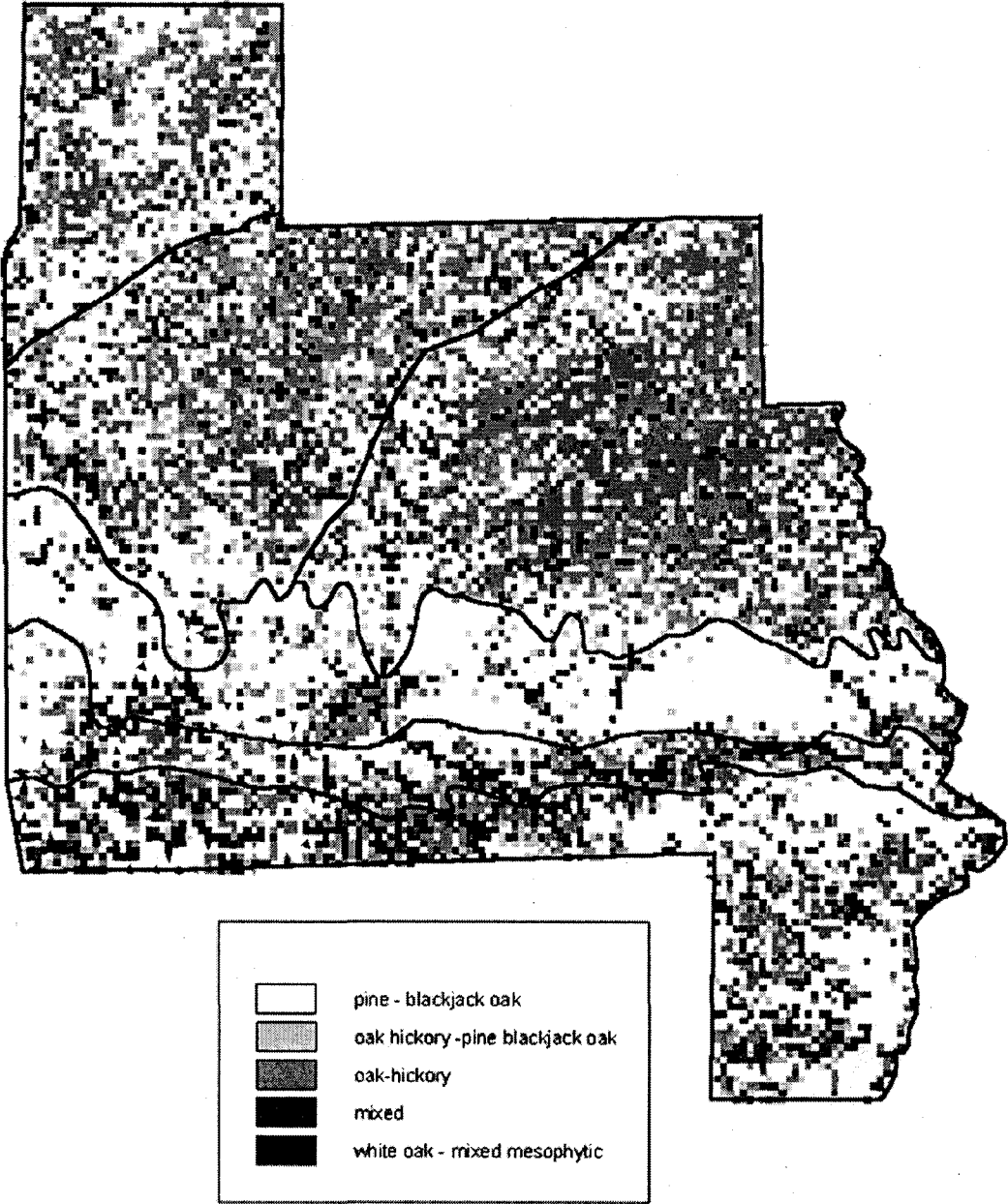
sected uplands of the Fall Line Hills and Red Hills (Figs. 1 and 5). On the other extreme, mixed mesophytic species (including ash, bay, beech, birch, cherry, elm, hackberry, holly, ironwood, magnolia, maple, and poplar) were most abundant in the broad valleys of the Red Hills, and to a lesser extent, in the narrower stream valleys of the other physiographic sections. White oak occurred on similar sites, but locations of its peak concentrations did not always coincide with those of mixed mesophytic species (Fig. 6). White oak and mixed mesophytic species only roughly overlapped in distribution. Likewise, post oak and hickory exhibited similar patterns of association with physiography, but in the Opelika Plateau where both were most abundant, hickory was concentrated in a much smaller region (Figs. 5 and 6).

Although landforms are roughly comparable among physiographic sections, their exact nature differs to some degree. For example, sideslopes are steeper in the Fall Line Hills and Coosa Valley than the Black Belt, and stream valleys are broad and level in the Red Hills and Black Belt, while they are steep and narrow in the Ashland and Opelika plateaus. Consequently, response of each species to landforms was generally consistent, but fluctuated somewhat throughout the study area (Figs. 7 and 8). This was true for both dominant species of the pine – blackjack oak community. Across all six physiographic sections, pine was negatively associated with stream valleys and positively associated with south sideslopes and hilltops. Magnitude of this relationship varied considerably, and was strongest in the Fall Line Hills. Patterns of landform association for blackjack oak were also fairly constant throughout the study area and overlapped to a large extent with those of pine. The major difference between these species was the lack of association between blackjack oak and hilltops (Figs. 7 and 8). In the white oak – mixed mesophytic community, landform associations were also quite consistent. All species in all physiographic sections were most strongly associated with stream valleys with the one exception that dogwood was associated with north sideslopes in the Red Hills. Other weaker associations include that of sourwood with north sideslopes in the Coosa Valley, Ashland Plateau, and Red Hills and white oak with north sideslopes in the Black Belt (Figs. 7 and 8).

Landform associations for species of the oak–hickory community were most variable among physiographic sections and among species. The only commonality was that all species of this community were rarely associated with south side slopes. Differences among species were evident in the Coosa Valley where post oak and hickory were most abundant on the valley floor; chestnut, chestnut oak, and black gum were most abundant on north sideslopes; while the red oaks were associated with the stream valleys and hilltops (Fig. 7). In the Piedmont, chestnut once again was associated with the north sideslopes, but post oak peaked in abundance on the hilltops, and peak abundance of the red oaks shifted from hilltops and stream valleys to hilltops and north sideslopes (Fig. 7). Black gum was associated with stream valleys, a pattern that also occurred throughout the Coastal Plain (Figs. 6 and 7). Only in the highly dissected Fall Line Hills were all other species of the oak–hickory community positively associated with stream valleys (Fig. 8). Farther south in the Black Belt and Red Hills, landform associations of oak–hickory species were similar to those in the



**Fig. 3.** Community map of east-central Alabama. The three major communities are oak–hickory, white oak – mixed mesopytic, and pine – blackjack oak. Oak – hickory – pine – blackjack oak is an equal combination of species from both communities. Mixed is an equal combination of species from the white oak – mixed mesopytic and pine – blackjack oak or oak–hickory community.



Piedmont. Exceptions include the strong association of hickory with north sideslopes and an association of chestnut with stream valleys in the Red Hills (Fig. 8).

**Witness tree irregularities**

Density analysis demonstrated that although witness tree concentration was uniform across large portions of east-central Alabama, concentration was unusually high along major rivers and unusually low in the southwestern corner of the study area (Fig. 9). Also, a mapping of red oak and black oak depicts sharp transitions in the abundance of these two species along township lines (Fig. 10). These transitions also occur on range lines (not shown). Thus, what one surveyor referred to as red oak, another referred to as black oak. For

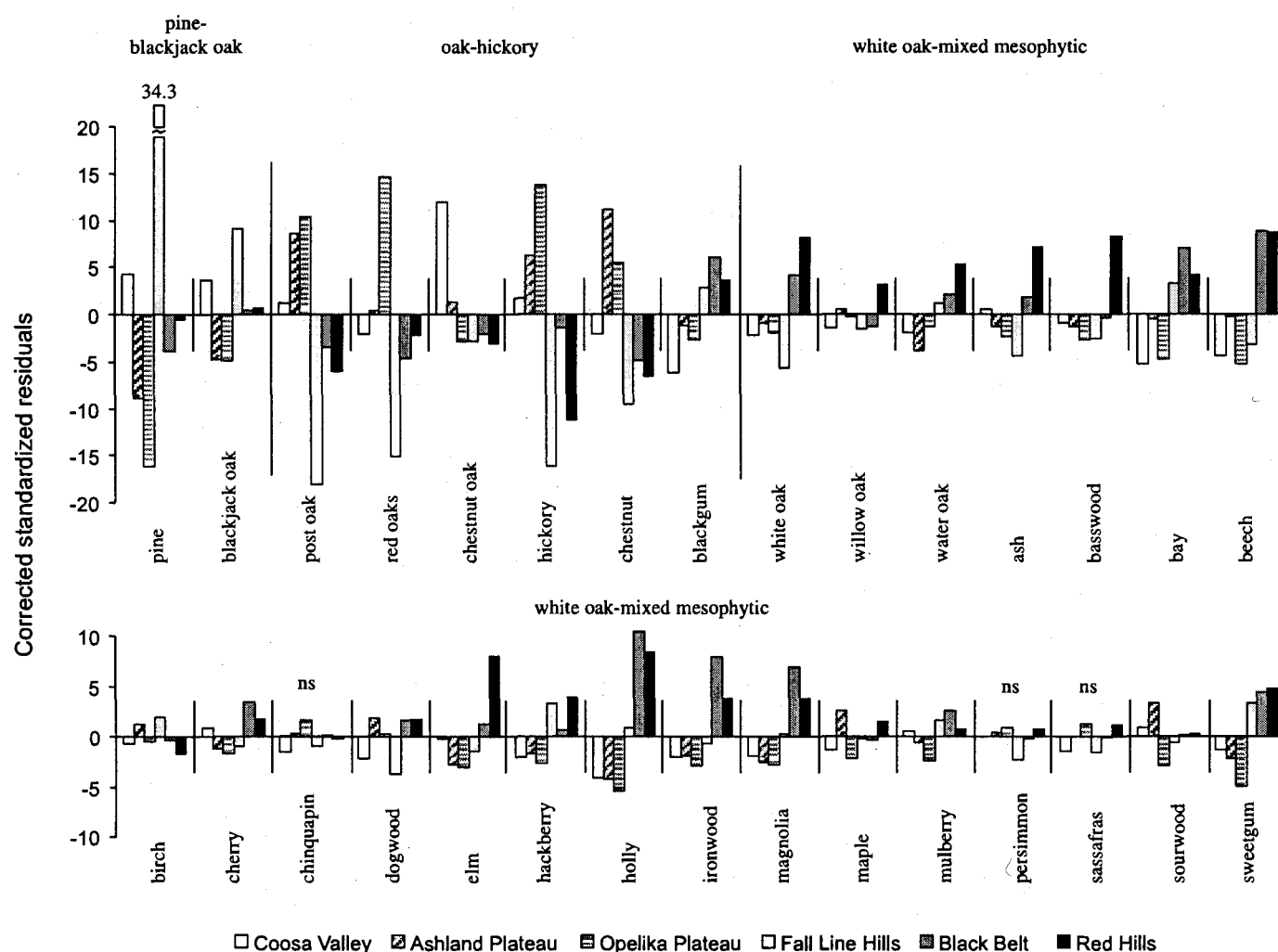
this reason, black oak, red oak, and the closely related Spanish oak were grouped into the category of “red oaks” in this study. Also, there were no significant differences between the frequencies of landforms at witness tree locations and landforms at half-mile grid intersections. Surveyors did not appear to have been biased toward selecting trees on specific landforms in the seemingly irregular surveys that occurred in the southwestern portion of the study area.

**Discussion**

Distribution of the pine – blackjack oak community in this study reflects its well-documented occurrence on upland, xeric sites (Garren 1943; Braun 1950; Shankman and Wills



**Fig. 4.** Associations of species to physiographic sections in east-central Alabama. A positive standardized residual indicates a positive association and a negative standardized residual indicates a negative association. ns, not significant ( $\alpha = 0.05$ ).



1995; Christensen 2000). Across the study area, both pine and blackjack oak were positively associated with south sideslopes and ridgetops and negatively associated with stream valleys. Given these very consistent species–site relationships and 80% dominance by pine, the composition of pine – blackjack oak appears to be relatively homogenous. However, the diversity of this community was probably much greater than that revealed by the witness tree record. Such low estimates of complexity are mainly due to the fact that surveyors failed to distinguish pine species.

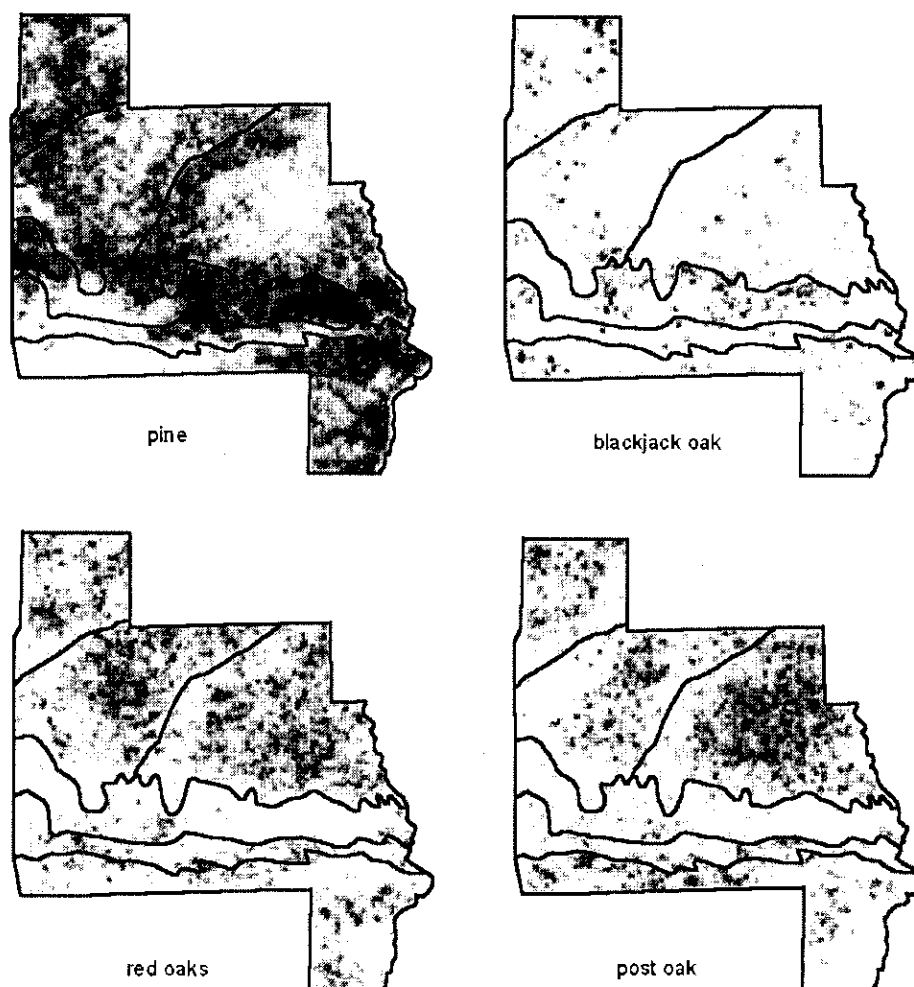
Several species including longleaf pine (*Pinus palustris*), shortleaf pine (*Pinus echinata*), loblolly pine (*Pinus taeda*), slash pine (*Pinus eliottii*), and Virginia pine (*Pinus virginiana*) likely occurred in the community; the density of each would have been dependent on local moisture and disturbance regimes (Braun 1950; Shankman and Wills 1995; Christensen 2000). Of all these species, longleaf is most competitive on sandy, dry, nutrient-poor sites prone to high fire frequencies, tending to occur on hilltops and upper south sideslopes (Mohr 1901; Braun 1950; Golden 1979; Burns and Honkala 1990; Christensen 2000). These forests are generally open and range from nearly pure longleaf pine to

two-story communities with a subcanopy dominated by blackjack oak, bluejack oak (*Quercus cinerea*), and turkey oak (*Quercus laevis*) (Mohr 1901; Braun 1950; Golden 1979; Christensen 2000). Virginia pine may have occurred on rock outcrops in the Piedmont and Ridge and Valley (Shankman and Wills 1995). On less extreme sites, shortleaf pine would have increased in dominance, canopy closure would have been greater, and oak and hickory would have become more common. Loblolly pine would have dominated on more mesic sites such as lower north sideslopes and coves (Braun 1950; Golden 1979; Burns and Honkala 1990).

Modern studies must also be used to clarify the composition and structure of the oak–hickory community. As was the case with pine, surveyors did not distinguish hickory to the species level, and although they attempted to distinguish red oak, black oak, and Spanish (southern red) oak, errors were clearly common. In the presettlement forests, southern red oak and mockernut hickory (*Carya tomentosa*) were probably most abundant on xeric sites, while black oak likely grew on a variety of upland sites, peaking in abundance under moderate to xeric conditions (Golden 1979). In stream valleys and coves, northern red oak (*Quercus rubra*), pignut



**Fig. 5.** Interpolations of major witness tree species in east-central Alabama. Dark shading indicates high concentrations of a species. Red oaks include red oak, black oak, and southern red oak.



hickory (*Carya glabra*), and bitternut hickory (*Carya ovata*) would have attained their highest frequencies (Golden 1979).

For those species that surveyors properly identified, other studies on modern and presettlement forests near and within the study area corroborate our species–site relationships (Golden 1979; Cowell 1995; Shankman and Wills 1995). All agree that post oak is associated with dry, upland sites. Also, others have noted chestnut oak on steep, north-facing sideslopes of the Ridge and Valley, Ashland Plateau, and to a lesser extent, on the Opelika Plateau (Golden 1979; Shankman and Wills 1995). Furthermore, the abundance of American chestnut on the Piedmont and ridges of the Ridge and Valley was consistent with its reputation of being positively associated with well-drained, acid loams and negatively associated with calcareous soils and those of the Coastal Plain (Russell 1987). Within the Piedmont, a province where American chestnut was abundant, previous witness tree studies report that American chestnut occurred on upland landforms, especially on north sideslopes (Cowell 1995; Shankman and Wills 1995). Our results indicate that degree of association with north sideslopes increased with topographic relief from the Opelika Plateau to the Ashland

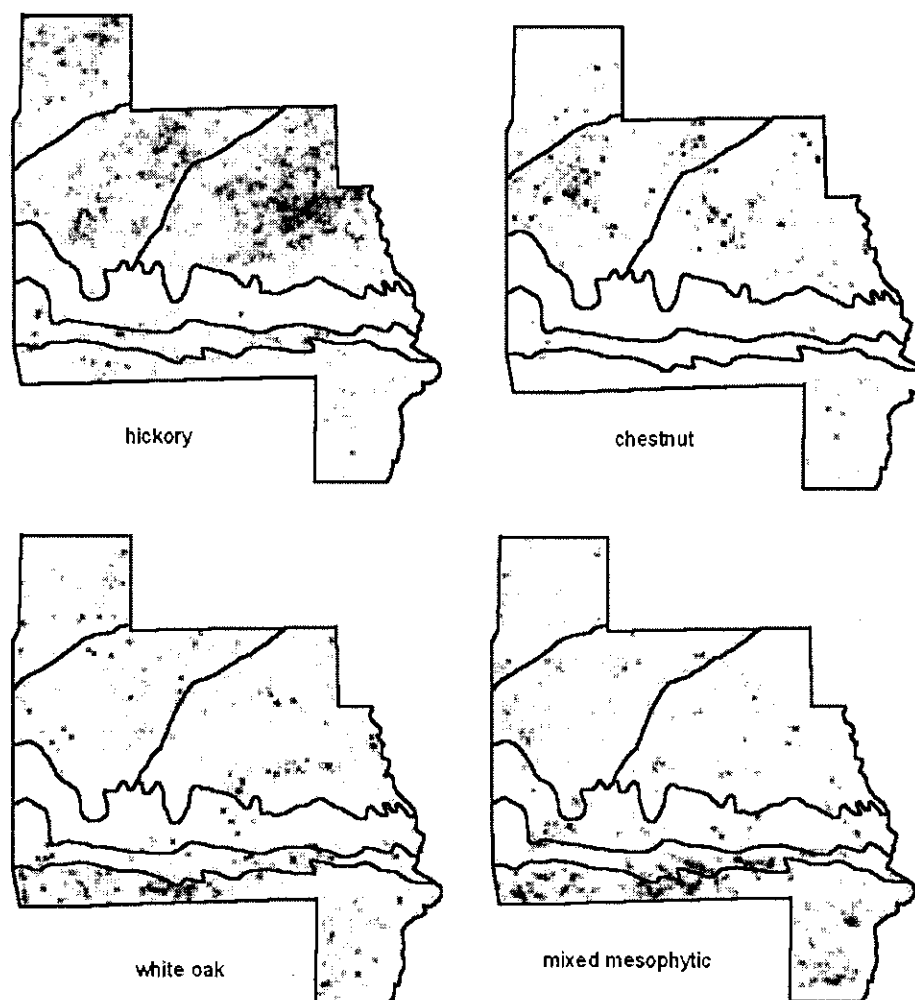
Plateau and Coosa Valley and that American chestnut tended to be associated with stream valleys where it occurred in the Coastal Plain.

Edaphic and topographic features also affected the structure of the oak–hickory community, particularly in the Black Belt. There, early traveler and surveyor accounts describe the oak dominated uplands as a combination of closed forests, oak savannas, and open grasslands. Soil reaction, which varies considerably across the Black Belt, accounts for a large part of this variability. Among upland sites, low-density forests dominated on alkaline clay soils, particularly in areas of moisture stress, while high-density forests occurred on acid loams (Jones and Patton 1966; Rankin and Davis 1971). Low tree densities were interpreted as open grasslands or savannas with scattered oaks and pines. In Montgomery County, over a third of the Black Belt was open grassland and only 10% supported high tree densities (Rankin and Davis 1971). Farther west in Sumter County, 23.4% of the Black Belt had low tree densities, including 13.4% classified as prairie (Jones and Patton 1966).

In the white oak – mixed mesophytic community, species were consistently associated with stream valleys (Figs. 6 and 7). However, wide dispersal in DCA axis 2 and slightly dif-



**Fig. 6.** Interpolations of major witness tree species in east-central Alabama. Dark shading indicates high concentrations of a species. Mixed mesophytic species are ash, bay, beech, birch, cherry, elm, hackberry, holly, ironwood, magnolia, maple, and poplar.



ferent patterns of association with physiography indicate that community composition was not uniform among stream valleys (Figs. 2 and 4). An analysis of modern forests in the Alabama Piedmont reveals similar groupings of mesophytic species and provides insight into community differences and the meaning of DCA axis 2 (Fig. 2) (Golden 1979). In his analysis of the Alabama Piedmont, Golden (1979) recognized three major mesophytic communities: sweetgum – water oak – red maple, small stream bottoms, and white oak. Sweetgum – water oak – red maple occurred in both wide and narrow stream valleys with poor drainage, while small stream bottom communities occurred in narrow, well-drained stream valleys (Golden 1979). Species characteristic of the sweetgum – water oak – red maple community have low scores for DCA axis 2 while species characteristic of the small stream-bottom community, including magnolia, holly, beech, and sourwood, have high scores. Indeed, plotting axis 2 scores in the GIS shows a clear concentration of high values in the broad valleys of the Coastal Plain and Black Belt. High scores also occur along the major river valleys throughout the study area.

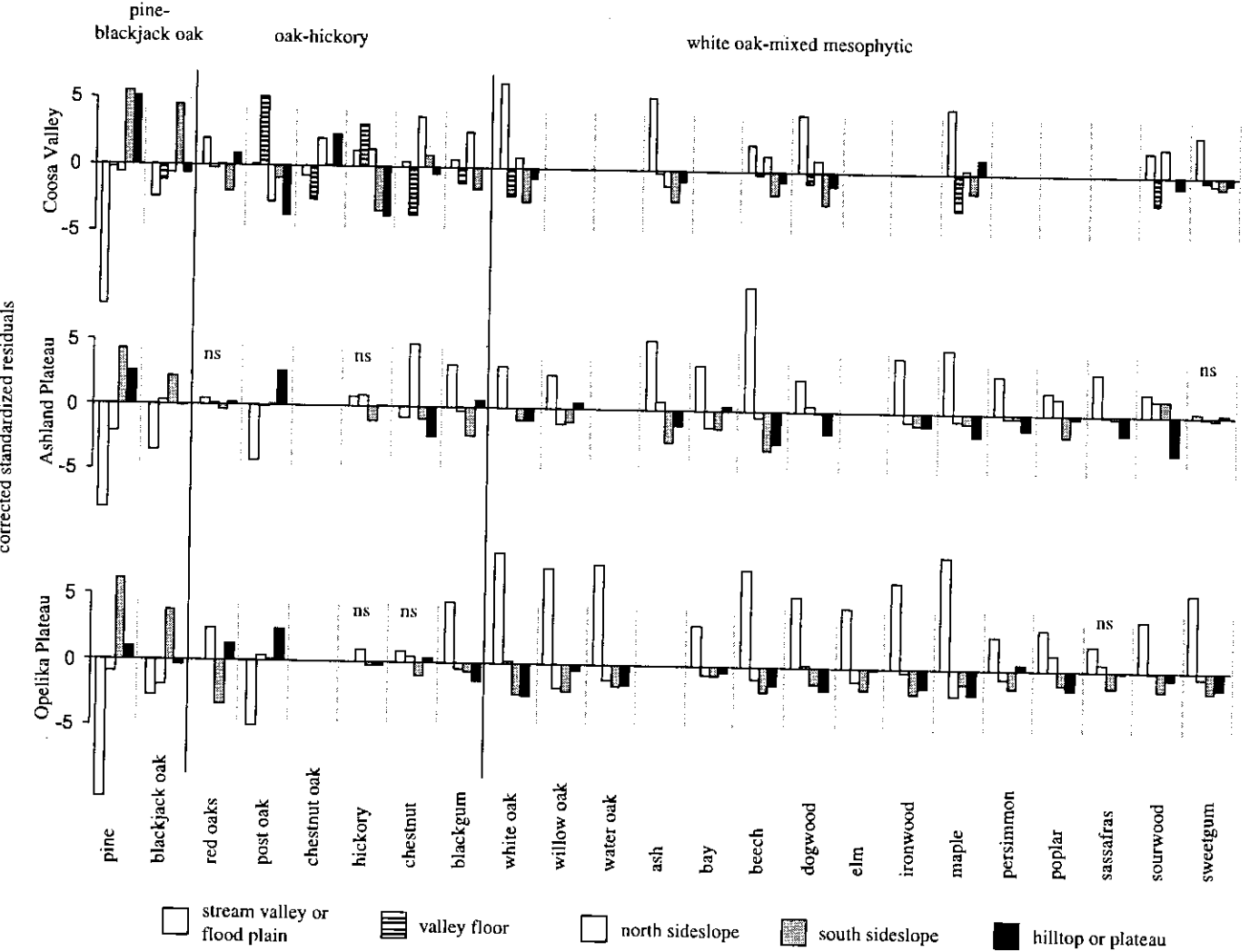
Golden's (1979) third mesophytic community, the white oak community, occurs on mesic upland sites and is dominated by white oak, shagbark and mockernut hickory, dog-

wood, tulip-tree, and sourwood. These species (with poplar most likely referring to tulip-tree in the witness tree record) cluster around white oak in the DCA plot (Fig. 2). They have the highest scores on axis 1, suggesting that of the mesophytic species, these are the most likely to occur on the uplands. Thus, it appears possible that this association not only occurs in the modern forest but could also be found in the presettlement forests.

While edaphic and topographic features can explain many patterns in structure and composition, disturbance undoubtedly influenced the presettlement forests. Evidence as to the importance of disturbance is the widespread dominance of pine, species of which are shade intolerant and require an open canopy to regenerate (Christensen 1989; Burns and Honkala 1990). Disturbances that could have opened the canopy and eliminated competition include hurricanes and smaller windstorms, yet these events were probably too localized or infrequent to account for such pervasive dominance (Cowell 1995). Instead, the most likely cause of pine's widespread distribution was periodic surface fires (Burns and Honkala 1990; Cowell 1995). Lightning ignitions were likely the most important ignition source in the pre-European settlement forests, particularly on upland flats and regions of low topographic relief. Also, early settlers and



**Fig. 7.** Associations of species to landforms in east-central Alabama. A positive standardized residual indicates a positive association and a negative standardized residual indicates a negative association. ns, not significant ( $\alpha = 0.05$ ). The Coosa Valley is within the Ridge and Valley province, and the Ashland and Opelika plateaus are within the Piedmont.



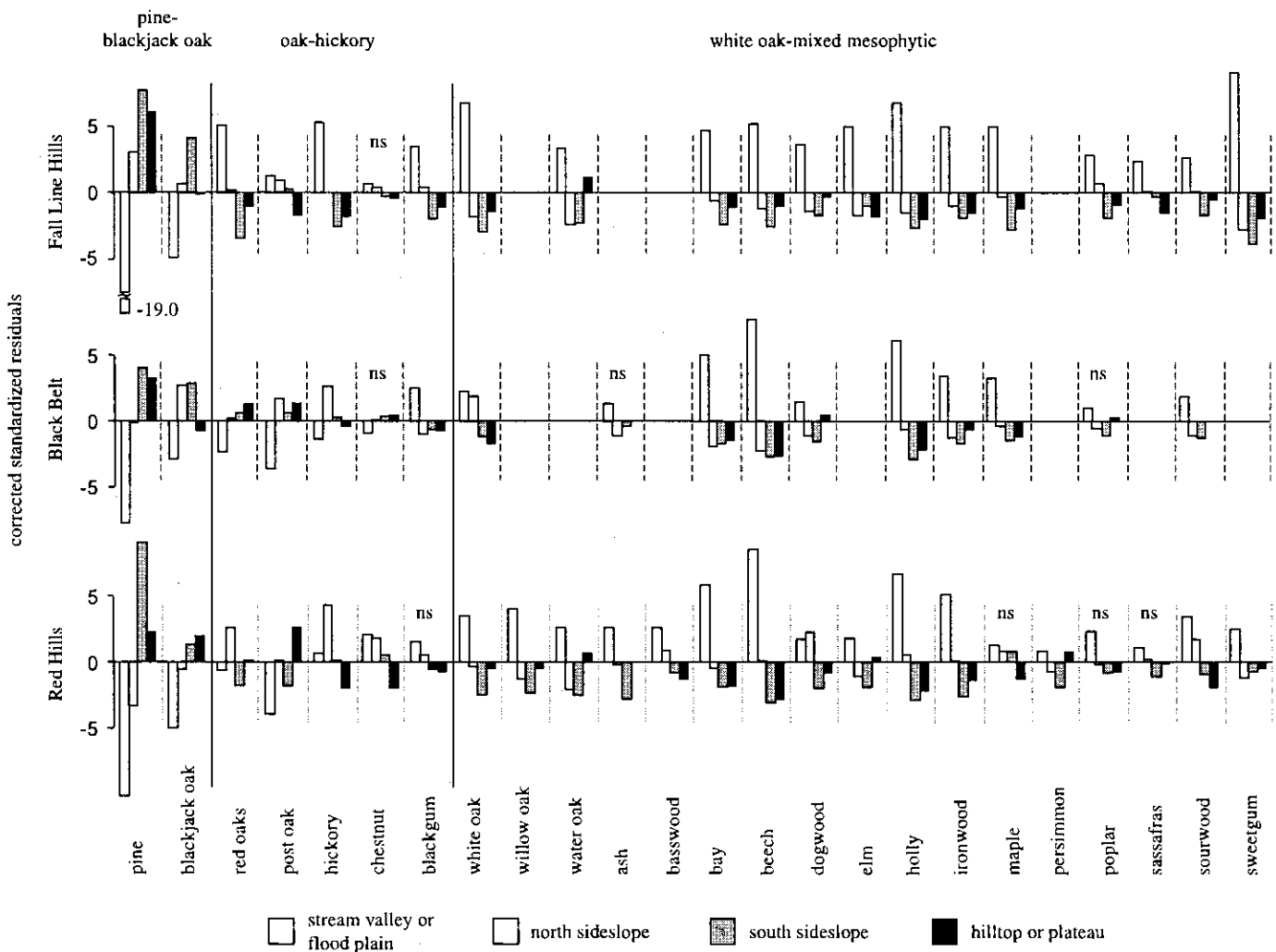
travelers provide numerous accounts of Native American burning. The influence of these anthropogenic fires would have been greatest in the most dissected portions of the Coastal Plain and Piedmont where the heterogeneity of the landscape would have prevented natural fires from spreading (Ware et al. 1993)

Contemporary studies suggest that recurring surface fires every 2 or 3 years would favor longleaf pine, while a fire interval of approximately 5 years would favor shortleaf pine, and an interval of at least 10 years would maintain loblolly pine (Pyne 1984). Oak and hickory tend to dominate sites with somewhat longer fire intervals, the shortest estimates of which range from 8 to 10 years (Cowell 1995; Shumway et al. 2001). Indeed, the loamy soils of the Piedmont and valley floors of the Coosa Valley where oak dominated would have inhibited fire compared with the sandy or thin soils of the Coastal Plain, mountains of the Ashland Plateau, and ridges of the Coosa Valley where pine dominated. Also, there may have been a fire-soil interaction on these sites that favored

oak, considering the partial overlap in the fire regimes between these genera. Relegation of fire-intolerant species, such as tulip-tree, ironwood, elm, and maple, to stream valleys is also consistent with the fire hypothesis. Furthermore, it is widely believed that without fire in southeastern forests, pine would be invaded or replaced by hardwood species (Oosting 1942; Garren 1943; Braun 1950; Quarterman and Keever 1962; Shankman and Wills 1995; Cowell 1995). For example, fire suppression in north-central Florida is causing a shift from pine to hardwoods on sandy upland sites (Delcourt and Delcourt 1977). A shift from pine to hardwoods was also noted in the lower Alabama Piedmont and is occurring at a much more rapid rate on moist sites than dry sites (Golden 1979). In an old loblolly pine forest on the Coastal Plain of Virginia, there was a shift towards increasing dominance of blackgum, sweetgum, and holly (*Ilex opaca* Ait.) (Abrams and Black 2000). Thus, hardwoods are capable of invading sites historically dominated by pine, but fire appears to have prevented such invasions.



**Fig. 8.** Associations of species to landforms in the Coastal Plain of east-central Alabama. A positive standardized residual indicates a positive association and a negative standardized residual indicates a negative association. ns, not significant ( $\alpha = 0.05$ ).



**Irregularities**

The fact that species–site relationships were strong, consistent, and in agreement with expected values suggests that surveyors identified trees with some degree of accuracy. The large number of tree species identified in the survey also supports this conclusion and indicates surveyors did not show strong biases toward any one species. However, caution should be used when interpreting the records of any two species that share many morphological characteristics. As was the case with red oak and black oak, surveyors may have had trouble distinguishing closely related species. No other species showed abrupt differences in abundance across township or range lines, yet it seems likely that turkey oak was included with blackjack oak. Almost certainly, turkey oak occurred in the study area and perhaps was even more abundant than blackjack oak in some places. However, because they are both low stature “scrub” oaks, surveyors may not have differentiated them.

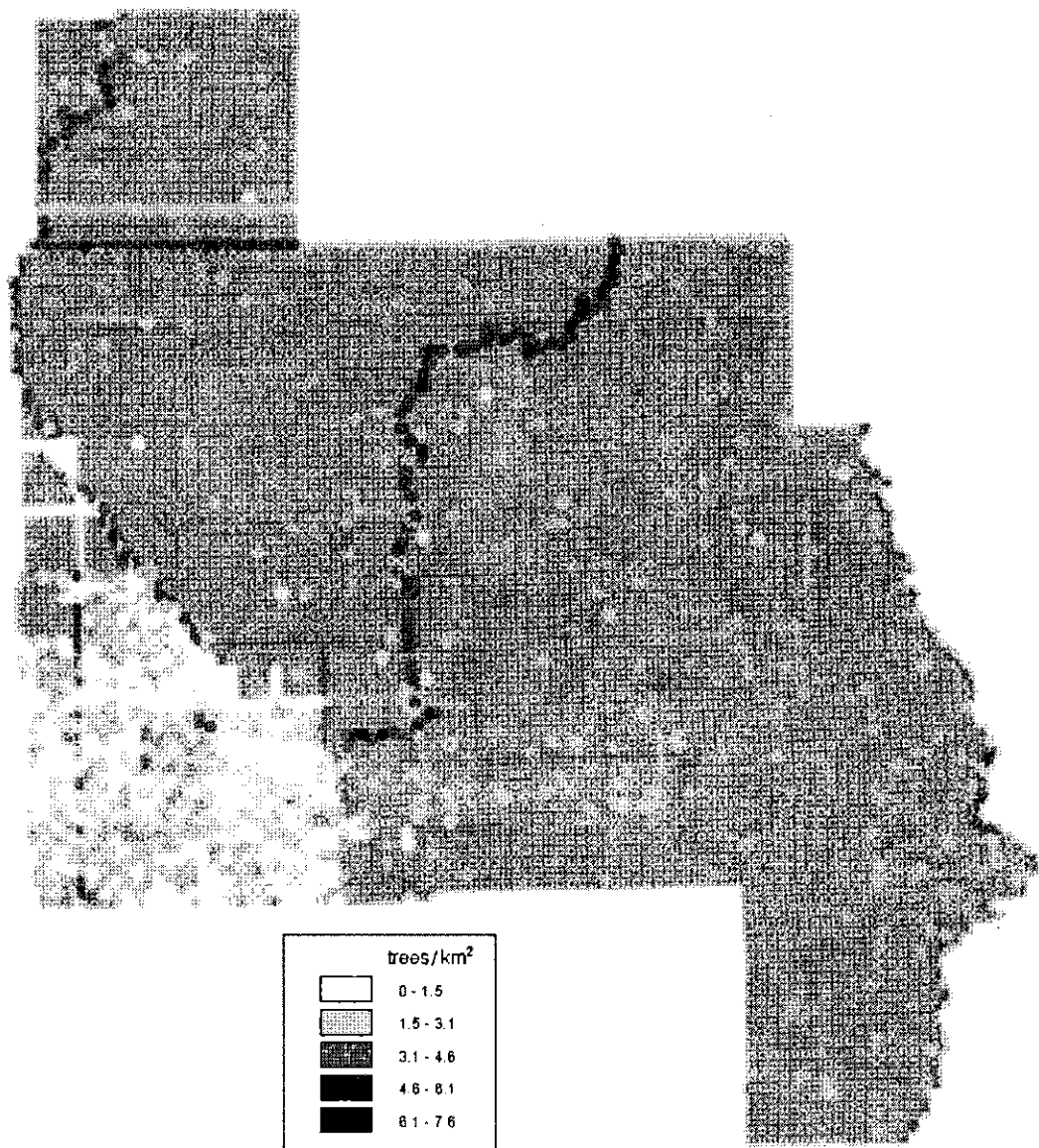
Irregular witness tree densities have been reported in metes and bounds witness tree data but not in Federal Public Land Survey data (Black and Abrams 2001). In the colonial (pre-1785) metes and bounds surveys that extended throughout New England and the mid-Atlantic (Black and Abrams 2001), surveys could be highly irregular in size and configu-

ration. Consequently, the density of witness trees (number of trees per unit area) may vary with the degree of topographic relief. For example, in Lancaster County, Pennsylvania, witness trees are much more concentrated in the mountainous uplands than the level lowlands (Black and Abrams 2001). Moreover, surveyors recorded significantly more trees on easily navigated landforms such as hilltops and stream valleys while avoiding the difficult terrain of sideslopes (Black and Abrams 2001). In east-central Alabama, the relatively high number of trees sampled along major river valleys could lead to a slight over-representation of flood plain species in the witness tree data. However, these increases in concentration are slight and probably do not affect the analysis of witness tree data to a significant extent in this study. As for other data sets, wherever surveyors were instructed to mark “wander points” at the borders of major water boundaries, such irregularities can be expected (Bourdo 1956; White 1991).

The low concentration of marked trees in the southwestern corner of the study area was more problematic. However, variations in witness tree density do not appear to correspond with any environmental variables. The most likely explanation for this irregularity is settlement of the region before surveys were completed. Montgomery, Ala., is lo-



Fig. 9. Density analysis of witness trees. Tree concentration is calculated as number of trees per square kilometre.



cated in this part of the study area and was incorporated in 1819 (Pickett 1851; Jones and Patton 1966). Indeed, surveyors frequently noted fields and clearings in this region (Jones and Patton 1966). Because these variations did not correlate with topography or geologic features, environmentally dependent analysis was unaffected. However, prior to performing interpolations, it was necessary to compute species' relative densities in square-kilometre blocks to compensate for the irregular tree density. Had these concentrations correlated with environmental variables, the study area would have to be subdivided to the level at which the variations occurred before species-site analyses or any interpolations could be conducted (Black and Abrams 2001).

Conclusions

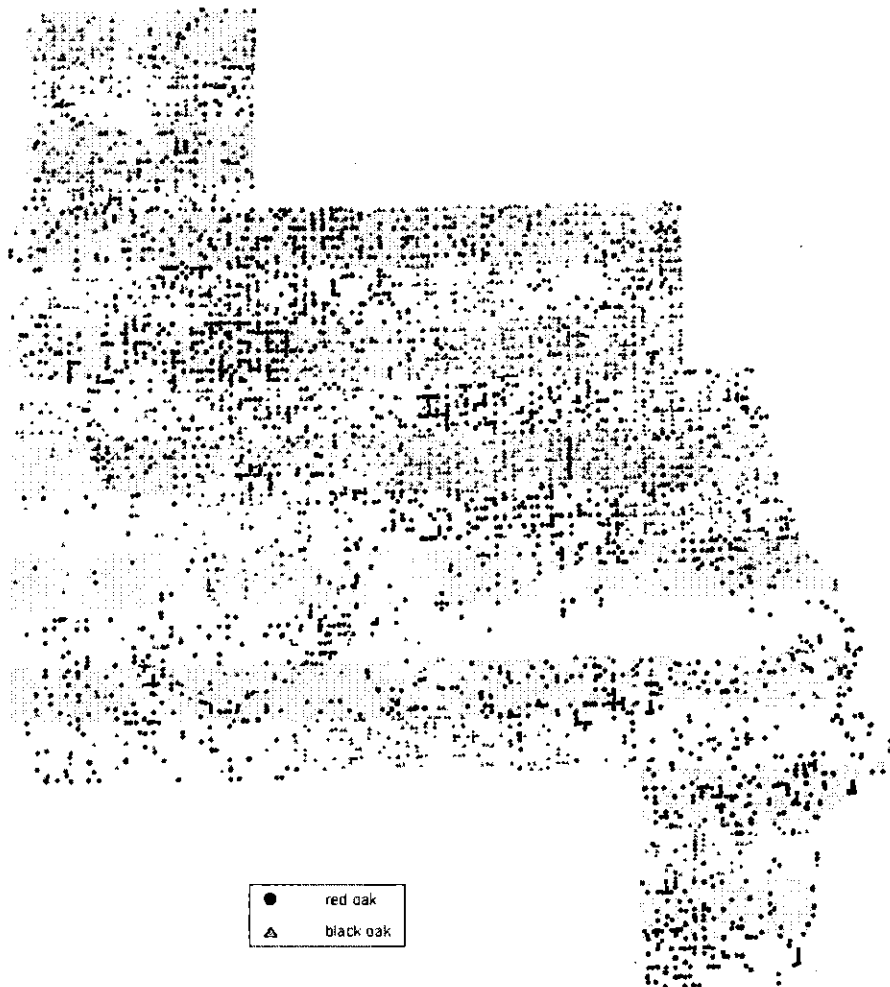
This study represents one of the largest pre-European settlement forest reconstructions in the southeastern United States. It provides a rare quantitative analysis of forest types that have been massively exploited and altered by post-

settlement land uses. This is especially true for the transitional forests between the Piedmont and Coastal Plain, which were never adequately described and are represented by only one remnant (Ware et al. 1993). This study also documents the relatively uniform sampling of witness trees in east-central Alabama, with the only major exception of a low tree density in the southwestern region of the study area. These irregularities did not correlate with any underlying environmental variables and were corrected by calculating the local relative density (i.e., within each square kilometre) of each species before interpolating. However, an irregularity that could not be corrected was the misidentification of red and black oak. The sharp differences in abundance of these two species across township lines suggests that closely related species might not be accurately identified in the witness tree record.

The use of environmentally independent and dependent analysis provided two different perspectives by which to describe presettlement forest composition. In this example, environmentally independent analysis was used to interpo-



**Fig. 10.** Map of red oak and black oak superimposed onto township lines. Abrupt changes in abundance of red oak and black oak occur across township lines, suggesting that surveyors did not accurately distinguish these two species.



late broad patterns in species abundance. Fine-scale variations were resolved using environmentally dependent analysis to describe species–landforms associations. When used alone, the scale of environmentally independent analysis is often too broad to capture important variations in forest composition, and environmentally dependent analysis assumes that edaphic and topographic variables are the sole regulators of vegetation. When combined, however, the shortcomings of both approaches are ameliorated. Resolution is maximized, and the effects of edaphic, topographic, and disturbance factors are all considered. Therefore, we suggest that both be applied to witness tree analysis, especially in regions where topography and geology are diverse.

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## Challenges in the development and use of ecological indicators

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### Abstract

Ecological indicators can be used to assess the condition of the environment, to provide an early warning signal of changes in the environment, or to diagnose the cause of an environmental problem. Ideally the suite of indicators should represent key information about structure, function, and composition of the ecological system. Three concerns hamper the use of ecological indicators as a resource management tool. (1) Monitoring programs often depend on a small number of indicators and fail to consider the full complexity of the ecological system. (2) Choice of ecological indicators is confounded in management programs that have vague long-term goals and objectives. (3) Management and monitoring programs often lack scientific rigor because of their failure to use a defined protocol for identifying ecological indicators. Thus, ecological indicators need to capture the complexities of the ecosystem yet remain simple enough to be easily and routinely monitored. Ecological indicators should meet the following criteria: be easily measured, be sensitive to stresses on the system, respond to stress in a predictable manner, be anticipatory, predict changes that can be averted by management actions, be integrative, have a known response to disturbances, anthropogenic stresses, and changes over time, and have low variability in response. The challenge is to derive a manageable set of indicators that together meet these criteria. Published by Elsevier Science Ltd.

**Keywords:** Ecological indicators; Hierarchy; Management resources

### 1. Introduction

As habitat fragmentation, changes in ecological condition, and loss of biodiversity escalate, society turns to science for guidance on dealing with complex environmental issues. Unfortunately, there are no simple solutions to many of the environmental problems of today, but clearly a commitment to con-

servation of natural resources and to understanding the implications of resource management and stress impacts is a necessary step towards addressing these complicated issues (Noss and Cooperrider, 1994). It is also imperative that, in connection with this focus on conservation, ecologists develop sound methods for monitoring, assessing, and managing ecological integrity through the use of indicators of ecological change. Ecological integrity refers to system wholeness, including the presence of appropriate species, populations, and communities and the occurrence of ecological processes at appropriate rates and scales (Angermeier and Karr, 1994; Karr, 1991) as well as the environmental conditions that support these taxa

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and processes. Thus, the concept of ecological integrity frames the selection of system-level indicators that are useful for resource managers (Karr, 1991).

Ecological indicators have several purposes (Cairns et al., 1993). They can be used to assess the condition of the environment or to monitor trends in condition over time. They can provide an early warning signal of changes in the environment, and they can be used to diagnose the cause of an environmental problem. The purpose influences the choice of ecological indicators. However, trade-offs between desirable features, costs, and feasibility often determine the choice of indicators.

A challenge in developing and using ecological indicators is determining which of the numerous measures of ecological systems characterize the entire system yet are simple enough to be effectively and efficiently monitored and modeled. Ecological indicators quantify the magnitude of stress, degree of exposure to the stresses, or degree of ecological response to the exposure (Hunsaker and Carpenter, 1990; Suter, 1993) and are intended to provide a simple and efficient method to examine the ecological composition, structure, and function of complex ecological systems (Karr, 1981) (e.g. see Table 1). The use of ecological indicators relies on the assumption that the presence or absence of, and fluctuations in, these indicators reflect changes taking place at vari-

ous levels in the ecological hierarchy, from genes to species and ultimately to entire regions (Noon et al., 1999).

The ecological hierarchy includes the functional, compositional, and structural elements that, when combined, define the ecological system and provide a means to select a suite of indicators representative of the key characteristics of the system (Fig. 1). All ecological systems have elements of composition and structure that arise through ecological processes. The characteristic conditions depend on sustaining key ecological functions which in turn, produce additional compositional and structural elements. If the linkages between underlying processes and composition and structural elements are broken, then sustainability and integrity are jeopardized and restoration may be difficult and complex.

Ideally the suite of indicators should represent key information about structure, function, and composition. The complexity of Fig. 1 only hints at the intricacy of the ecological system on which it is based. The series of nested triangles in the figure are meant to suggest that knowledge of one part of the triangle may provide information to the other aspects of the system. For example, often it is easier to measure structural features that can convey information about the composition or functioning of the system than to measure composition or function (Lindenmayer et al., 2000). Sometimes measures from one scale can provide information relevant to another scale. For example, the size of the largest patch of a habitat often restricts the species or trophic levels of animals that are able to be supported based solely on their minimal territory size (Dale et al., 1994). Even so, it is often difficult to know how large an area or how long to monitor (Dawe et al., 2000). The ecological system can be viewed as a moving target (Walters and Holling, 1990) with many system variables changing slowly and not stabilizing for a long time.

Table 1  
Example components and indicators for ecological integrity

| Hierarchy  | Processes   | Suggested indicators  |
|------------|---|---|
| Organism   | Environmental toxicity<br>Mutagenesis                           | Physical deformation<br>Lesions<br>Parasite load                                |
| Species    | Range expansion or contraction<br>Extinction                    | Range size<br>Number of populations   |
| Population | Abundance fluctuation<br>Colonization or extinction             | Age or size structure<br>Dispersal behavior                                     |
| Ecosystem  | Competitive exclusion<br>Predation or parasitism<br>Energy flow | Species richness<br>Species evenness<br>Number of trophic levels                |
| Landscape  | Disturbance<br>Succession                                       | Fragmentation<br>Spatial distribution of communities<br>Persistence of habitats |

## 2. Concerns and challenges

Although few scientists deny the benefits that indicators provide to research and management efforts, three concerns hamper the use of ecological indicators as a resource management tool (Landres et al., 1988; Kelly and Harwell, 1990; Noss, 1990;



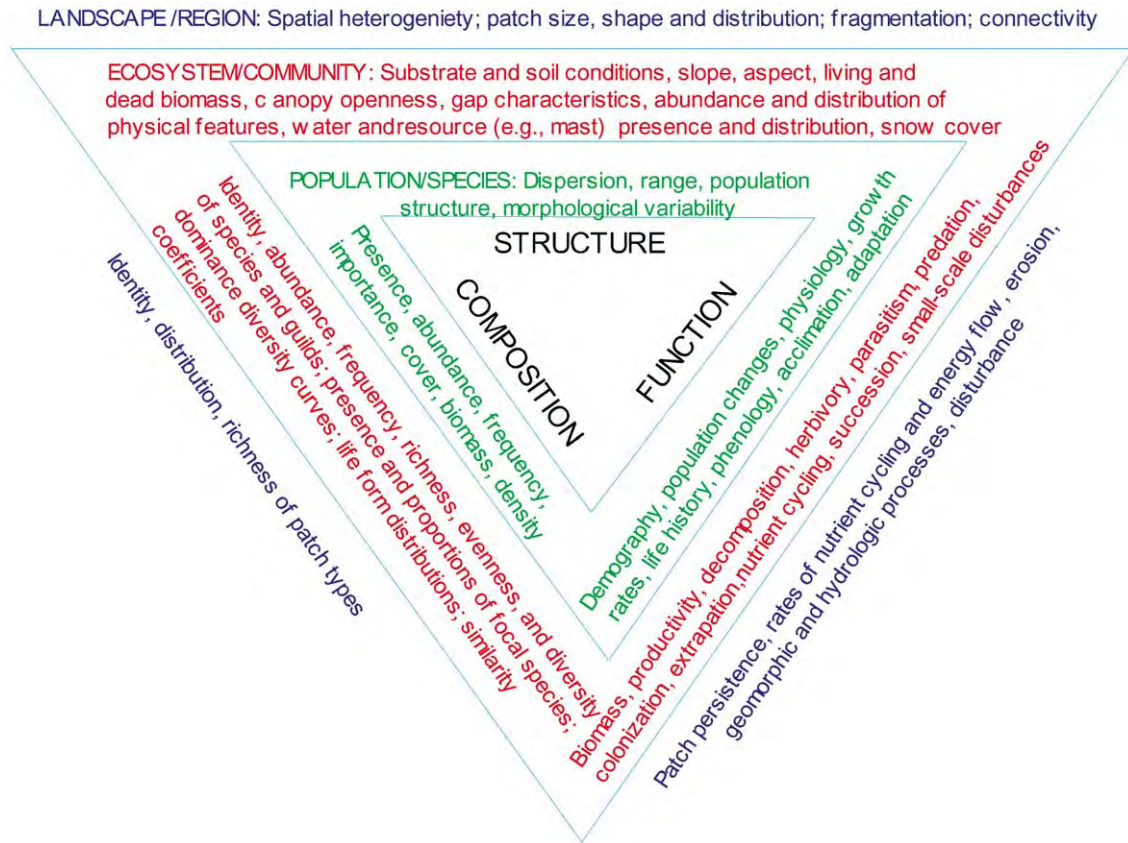


Fig. 1. The ecological hierarchy: a triangular representation of the key characteristics of composition, structure and function (derived from Franklin, 1988 and Noss, 1990).

Kremen, 1992; Cairns et al., 1993; Mills et al., 1993; Noss and Cooperrider, 1994; Gurney et al., 1995; Simberloff, 1997).

*2.1. Monitoring programs often depend on a small number of indicators and, as a consequence, fail to consider the full complexity of the ecological system*

By selecting only one or a few indicators, the focus of the ecological management program becomes narrow, and an oversimplified understanding of the spatial and temporal interactions is promoted. This simplification often leads to poorly informed management decisions. Indicators should be selected from multiple levels in the ecological hierarchy in order to effectively monitor the multiple levels of complexity within an

ecological system. Thus, a key challenge is to find a mix of measures which give interpretable signals, can be used to track the ecological conditions at reasonable cost, and cover the spectrum of ecological variation.

*2.2. Choice of ecological indicators is often confounded in management programs that have vague long-term goals and objectives*

Unclear or ambivalent goals and objectives can lead to “the wrong variables being measured in the wrong place at the wrong time with poor precision or reliability” (Noss and Cooperrider, 1994). Primary goals and objectives should be determined early in the process in order to focus monitoring on current and future management issues. Ecological indicators can



Table 2  
Criteria for ecological indicators

---

|   |
|---|
| Are easily measured   |
| Are sensitive to stresses on system   |
| Respond to stress in a predictable manner   |
| Are anticipatory: signify an impending change in the ecological system  |
| Predict changes that can be averted by management actions   |
| Are integrative: the full suite of indicators provides a measure of coverage of the key gradients across the ecological systems (e.g. soils, vegetation types, temperature, etc.) |
| Have a known response to natural disturbances, anthropogenic stresses, and changes over time  |
| Have low variability in response  |

---

then be selected to measure system characteristics that most closely relate to those management concerns. However, society typically has selected resource management goals concerned solely with short-term profit (e.g. maximum crop yield in agricultural systems or maximum timber production in forests). These goals may jeopardize the long-term maintenance of healthy ecological systems. Management goals, and thus indicator selection should be tied to an understanding of both the short-term and long-term consequences of resource management decisions.

*2.3. Management and monitoring programs often lack scientific rigor because of their failure to use a defined protocol for identifying ecological indicators*

Lack of robust procedures for selecting ecological indicators makes it difficult to validate the information provided by those indicators. Until standard methods are established for selecting and using indicators, interpretation of their change through space and time remains speculative (Noss, 1999). The creation and use of standard procedures for the selection of ecological indicators allow repeatability, avoid bias, and impose discipline upon the selection process, ensuring that the selection of ecological indicators encompasses management concerns (Slocombe, 1998; Belnap, 1998).

### 3. Criteria for selecting ecological indicators

Selection of effective indicators is key to the overall success of any monitoring program. In general, ecological indicators need to capture the complexities of

the ecosystem yet remain simple enough to be easily and routinely monitored. In order to define ecological indicators, however, it is first necessary to set forth criteria used to select potential ecological indicators. Building upon discussions by Landres et al. (1988), Kelly and Harwell (1990), Cairns et al. (1993), and Lorenz et al. (1999), we suggest that ecological indicators should meet the following criteria (Table 2):

- *Be easily measured:* The indicator should be straightforward and relatively inexpensive to measure. The metric needs to be easy to understand, simple to apply, and provide information to managers and policymakers that is relevant, scientifically sound, easily documented, and cost-effective (Stork et al., 1997; Lorenz et al., 1999). Historically, canaries were carried into mines to warn workers of the presence of methane and other gases that can lead to an explosion. The death of a canary is an easily observed, if unfortunate (for the canary), result of the presence of volatile gases. When a bird succumbed to toxic gas, it was an indication to the miners were in imminent danger.
- *Be sensitive to stresses on the system:* The ideal ecological indicator is responsive to stresses placed on the system by human actions while also having limited and documented sensitivity to natural variation (Karr, 1991). While some indicators may respond to all dramatic changes in the system, the most useful indicator is one that displays high sensitivity to a particular and, perhaps, subtle stress, thereby serving as an early indicator of reduced system integrity. For example, the gopher tortoise (*Gopherus polyphemus*) is highly sensitive to soil disturbances, and their absence in otherwise suitable sites suggests past physical disturbances. (This



Table 3

An example of an ecological indicator

An example of an ecological indicator is the presence of the cyanobacteria *Oscillatoria rubescens* in lakes that are on the verge of extreme eutrophication. The role of this cyanobacteria as an indicator was first identified in Lake Washington (Edmondson and Lehman, 1981; Edmondson, 1991). In the first half of the 20th century, metropolitan Seattle discharged treated sewage high in phosphorus content into Lake Washington. By 1955, effluent contributed more than 50% of the total phosphorus input into the lake. Increased nutrient levels altered lake productivity and resulted in massive blooms of cyanobacteria that negatively affected fish populations and greatly reduced water clarity. Public attention was called to the problem, and the resulting reversal of this eutrophication process occurred when sewage was diverted from the lake and into Puget Sound. The resulting drop in nutrient additions eliminated algal blooms and increased water clarity. Now, *O. rubescens* is used as an indicator of impending eutrophication worldwide. It satisfies three elements of an ecological indicator in that it is easily measured, it signifies an impending change in the ecosystem, and both the potential ecosystem change and the high level of the indicator can be averted by management action. (Unfortunately, Puget Sound still suffered even after advances were made in the sewage treatment system).

interpretation of the tortoise's absence in sand hills areas can be made only when there are no other pressures on the tortoise, such as harvesting tortoise for food or gassing burrows to collect snakes.)

- *Respond to stress in a predictable manner:* The indicator response should be unambiguous and predictable even if the indicator responds to the stress by a gradual change (such as the increase in density of the cyanobacteria *Oscillatoria rubescens* in polluted lakes). Ideally, there is some threshold response level at which the observable response occurs before the level of concern (Table 3).
- *Be anticipatory, i.e. signify an impending change in key characteristics of the ecological system:* Change in the indicator should be measurable before substantial change in ecological system integrity occurs. For the canaries in the coal mine example, the birds died at levels of toxic gases not quite sufficient to create an explosion or be toxic to humans.
- *Predict changes that can be averted by management actions:* The value of the indicator depends on its relationship to possible changes in management actions. For example, the presence of young longleaf pine (*Pinus palustris*) serves as a measure of the recurrence of fire at Eglin Air Force Base (AFB) on the Florida panhandle (McCay, 2000). With fire suppression, the normally restricted distribution of sand pine (*P. clausa*) expanded from 2400 ha to over 24,000 ha, and young longleaf pine became rare. It has only been through the reintroduction of a regular fire regime at Eglin AFB that the historically dominant species, longleaf pine, has been reestablished. Today, the presence of young longleaf pine, which survive light fire, is a result of the artificially-induced 2 to 3-year fire regimes that occur not only at Eglin AFB but also at other managed areas in the southeastern United States. In contrast, effects of large, infrequent disturbances often serve as a counter example of changes that cannot be averted by management actions (Dale et al., 1998). Ecological effects of volcanoes, large climate-induced fires, and hurricanes cannot be predicted by ecological indicators nor deterred.
- *Are integrative: the full suite of indicators provides a measure of coverage of the key gradients across the ecological systems (e.g. gradients across soils, vegetation types, temperature, space, time, etc.):* The full suite of indicators for a site should integrate across key environmental gradients. For example, no single indicator is applicable across all spatial scales of concern. Brooks et al. (1998) developed a suite of indicators for forested riparian ecosystems of Louisiana that behave predictably across scales and can be aggregated to provide an assessment of the entire system. In a like manner, the ability of the suite of indicators to embody the diversity in soils, topography, disturbance regimes, and other environmental gradients at a site should be considered.
- *Have a known response to disturbances, anthropogenic stresses, and changes over time:* The indicator should have a well-documented reaction to both natural disturbance and to anthropogenic stresses in the system. This criterion would pertain to conditions that have been extensively studied and have a clearly established pattern of response. Focal species are often the only types of species that



Table 4  
Categories of focal species

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|   |
|---|
| <i>Indicator species</i> are species whose status is indicative of the status of a larger functional group of species, reflects the status of key habitats, or acts as an early warning to the action of an anticipated stressor (e.g. white-tailed deer ( <i>Odocoileus virginianus</i> ) populations that signify the availability of forest-grassland margins).                |
| <i>Keystone species</i> have much greater effects on one or more ecological processes than would be predicted from their abundance or biomass alone (Power et al., 1996) (e.g. the red-cockaded woodpecker ( <i>Picoides borealis</i> ) creates cavities in living trees that provide shelter for 23 other species (Dennis, 1971)).   |
| <i>Ecological engineers</i> alter the habitat to their own needs and by doing so affect the fates and opportunities of other species (e.g. Jones et al., 1994; Naiman and Rogers, 1997) (such as the gopher tortoise ( <i>G. polyphemus</i> ) that digs burrows used by many other species or the beaver ( <i>Castor canadensis</i> ) whose dams create wetlands).                |
| <i>Umbrella species</i> have either large area requirements or use multiple habitats that encompass the habitat requirements of many other species (e.g. the northern spotted owl ( <i>Strix occidentalis caurina</i> ) that occupy old growth forest in the Pacific Northwest).  |
| <i>Link species</i> play critical roles in the transfer of matter and energy across trophic levels or provide a critical link for energy transfer within complex food webs. For example, prairie dogs ( <i>Cynomys</i> spp.) in grassland ecosystems convert primary plant productivity into animal biomass. Prairie dog biomass, in turn, supports a diverse predator community. |
| <i>Special interest species</i> include threatened and endangered species, game species, charismatic species and those that are vulnerable due to their rarity.   |

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have a large enough foundation of information to indicate long-term trends and responses to change (Table 4). Landscape structure can also respond to human disturbances versus natural disturbances in a predictable manner (Krummel et al., 1987). Thus, landscape metrics can serve as useful indicators of change (O'Neill et al., 1988; Dale and Pearson, 1997; Gustafson, 1998; Klemas, 2001).

- *Have low variability in response:* Indicators that have a small range in response to particular stresses allow for changes in the response value to be better distinguished from background variability. As a counter example, seabirds were a poor indicator of the ecological cost of the 1989 Exxon Valdez oil spill and of the benefit of subsequent steam cleaning. More than 30,000 oiled bird carcasses were retrieved following the spill, but because of the high variability inherent in seabird populations, the population dynamics of birds in the spill areas are difficult to distinguish from the population dynamics of birds at nonspill sites (Wiens, 1996).

A challenge is to derive a manageable set of indicators that together meet these criteria.

#### 4. Conclusions

Ecological indicators are used to monitor, assess, and manage natural resources. A difficulty in selecting

appropriate indicators is dealing with the complexity of ecological systems. Thus, it is necessary to use a suite of indicators representative of the structure, function, and composition of ecological systems. The need to communicate the scientific concepts of ecological indicators to non-scientists is being tackled by teams of environmental scientists working with social scientists (e.g. Schiller et al., 2001). Yet, integrating ecological indicators with social and economic goals for resource management remains a big challenge.

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## Section 6: Landscape Indicators

This section consists of three scientific papers that detailed the use of witness trees and the selection of landscape indicators.

Appendix A: Foster, Thomas, Bryan Black, and Marc Abrams. 2004. A witness tree analysis of the effects of Native American Indians on the pre-European settlement forests in east central Alabama. **Human Ecology** 32 (1).

Appendix B: Olsen, L.M., Dale, V.H. and H.T. Foster. In press. Landscape patterns as indicators of ecological change at Fort Benning, GA. **Land Use and Urban Planning**.

Appendix C: Olsen, L.M., Washington-Allen, R.A, and V.H. Dale. Resubmitted in response to “accepted with revision.” Using Landscape Metrics to Detect Changes over 173 Years at Fort Benning,GA, USA. **GIScience and Remote Sensing**



# **A Witness Tree Analysis of the Effects of Native American Indians on the Pre-European Settlement Forests in East-Central Alabama**

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## **Abstract**

Witness tree data from the southeastern United States (lat 33°30' N, long 86°30' W) were analyzed using catchment and distance analysis to quantify the effects of Native American settlement on the composition of forest trees. Thirty Creek Indian villages comprising 18 settlement catchments were included in the sample, which is the largest Native American–forest interaction study using witness trees to date. Lower frequencies of *Pinus* spp. were observed within village catchments of the Coastal Plain and Ridge and Valley. Elevated frequencies of early succession species were observed surrounding 2 km village catchments. Distance analysis at two relatively isolated towns showed that *Pinus* increases in frequency beyond 2000 m from villages while *Carya* had the opposite result. Field and fruit species were more frequent within 6000 m of villages and then dropped off in frequency. Fire-sensitive tree species appear to be in a spatially cyclical pattern.

## **Keywords**

Native American, forest composition, witness tree, human–forest interaction, Alabama



**Landscape Patterns as Indicators of Ecological Change**  
**at Fort Benning, Georgia, USA**

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## **Abstract**

This research examined landscape indicators that signal ecological change in both intensely used and lightly used lands at Fort Benning, Georgia. Changes in patterns of land cover through time affect the ecological system by altering the proportion and distribution of habitats for species that these cover types support. Landscape patterns, therefore, are important indicators of land-use impacts, past and present, upon the landscape. This analysis of landscape pattern began with a landscape characterization based on witness tree data from 1827 and the 1830s and remotely sensed data from 1974, 1983, 1991, and 1999. The data from the early 1800s, although coarse, were useful in characterizing the historical range of variability in ecological conditions for the area. The steps for the analysis involved the creation of a land-cover database and a time series of land cover maps, computation of landscape metrics, and evaluation of changes in those metrics over time as evidenced in the land-cover maps. We focused on five cover types (bare/developed land, deciduous forest, mixed forest, pine forest, and nonforest vegetated land), for they reveal information important to resources management at Fort Benning. An examination of land-cover class and landscape metrics, computed from the maps, indicated that a suite of metrics adequately describes the changing landscape at Fort Benning, Georgia. The most appropriate metrics were percent cover, total edge (km), number of patches, descriptors of patch area, nearest neighbor distance, the mean perimeter-to-area ratio, shape range, and clumpiness. Identification of such ecological indicators is an important component of building an effective environmental monitoring system.



**Keywords:** Endangered species; Habitat; Land-cover change; Land use, Landscape metrics; Presettlement vegetation; Pine forests; Remote sensing; Witness trees.

## 1. Introduction

Identifying ecological indicators is an important component in describing an ecological system, establishing potential metrics of change, and building an effective environmental monitoring system. Together the suite of indicators should represent the range of ecological conditions in the ecological system, serve as signals of environmental change, and be simple enough to allow cost-effective monitoring and modeling (Hunsaker and Carpenter, 1990; Kelly and Harwell, 1990; Noss, 1990; Cairns et al., 1993; Dale and Beyeler, 2001). Landscape metrics that include quantifiable measures of landscape fragmentation have been developed to capture important aspects of landscape pattern in a few numbers (O'Neill et al., 1988; Riitters et al., 1995; Turner et al., 2001). These numbers can often be correlated with land-use change and ecological processes. By using such metrics to examine and quantify landscape patterns through time, researchers may determine the long-term impacts of previous land use (e.g., Bürgi, 1999; Griffith et al., in press). Landscape metrics can thus capture changes in pattern, be implemented along a variety of spatial scales, and be useful indicators of land-cover changes due to prior land use and management.

Landscape metrics should be useful to any landowner or manager interested in knowing whether the landscape pattern is changing and how. Such metrics could be



particularly useful to the U.S. Department of Defense (DoD), which has taken a proactive approach to land management. A major component of DoD's mission is to provide adequate lands for military training and operations. Military lands are subject to the same environmental responsibilities and regulations as other federal lands, and they contain a high density of federally and state listed species (Leslie et al., 1996), which by law require protection. Given the nature of military land use, military testing and training may degrade or fragment critical habitats and put species at risk. Vegetation and habitat play a crucial role in military training by providing concealment and an element of realism in training exercises. By harming the vegetation and habitat, military testing and training may degrade the very landscape characteristics that are necessary for thorough training and ultimately compromise the ability of a military installation to fulfill its mission, either by jeopardizing realism or by violating the provisions of the Endangered Species Act.

As part of DoD's proactive approach to land management, environmental monitoring and management plans are being developed to assist military installations in balancing their training requirements and environmental responsibilities in both the short term and long term (e.g., Diersing et al., 1992). The immediate goal of our research was to identify and map trends in land-cover change that have occurred since 1827 at Fort Benning, Georgia, and to develop techniques to measure those trends. This analysis is a multi-step process beginning with the creation of land-cover maps for different time periods and culminating with the computation, summarization, and evaluation of landscape metrics.

Specifically, we examined trends in landscape metrics that relate to changes in ecological conditions over time such as changes in vegetation type and pattern. This



analysis identified landscape metrics that are useful indicators of change at Fort Benning. An indicator should adequately characterize an aspect of the system and be able to be implemented for management purposes (Dale and Beyeler, 2001). Although Fort Benning was the focal site of this project, the ultimate goal was to develop an approach for identifying landscape indicators that would be useful for a diversity of locations and management types. This wider effort is ongoing, and the results published here include a description of the methods and tools used to create the land-cover time series and to compute landscape metrics. Only a short discussion of the metrics is provided because other references discuss them in depth (O'Neill et al., 1988; Hunsaker and Carpenter, 1990; Noss, 1990; Gustafson, 1998; Turner et al., 2001).

How does land cover change over a long term? It is important to address this question in order to understand current land-cover patterns as well as the historical range of variability (HRV). Land-cover patterns are a function of prior conditions, but how these conditions affect current vegetation cover is still a critical area of ecological research (Wu and Hobbs, 2002). Knowledge about natural variability provides a framework for improving knowledge about ecological systems, for understanding how human actions modify the landscape, and for evaluating consequences of proposed management actions (Landres et al., 1999). Clearly, HRV requires a long-term perspective (Brown et al., 2000); the appropriate means for obtaining information about landscape changes over time are still being explored (e.g., see Strittholt and Boerner, 1995; Grissino-Mayer, 1999; Hessburg et al., 1999; Kepner, 2000; Keane et al., 2002; Tinker et al., 2003). A variety of methods are available to analyze changes over time (Lu et al., 2004). In this study, witness tree data from historical land surveys provide a means



to estimate the vegetation conditions prior to European settlement (Delcourt and Delcourt, 1996; Cogbill, 2000; Manies and Mladenoff, 2000; Brewer, 2001; Dyer, 2001; Manies et al., 2001; Cogbill et al., 2002; Leadbitter et al., 2002; Schulte et al., 2002), and remotely sensed land-cover data provide a means to estimate vegetation in recent decades. This paper analyzes data collected over a 172-year period and uses the data to determine changes in land-cover patterns in the area of Fort Benning since the early nineteenth century.

## **2. Site description and history**

Fort Benning is a military base located southwest of Columbus, Georgia. Most of the installation lies in west central Georgia, but a small part extends into Russell County, Alabama (Fig. 1). The climate is characterized by long, hot summers and mild winters. Precipitation is regular throughout the year, with most occurring in the spring and summer (Knowles and Davo, 1997). The installation is situated on the Fall Line transition zone, which is the geographic area between the Southern Appalachian Piedmont and the Coastal Plain. Soils are composed of clay beds, weathered Coastal Plain material, and alluvial deposits from the Piedmont (Knowles and Davo, 1997). The base is 73,503 hectares in size.

Prior to establishment of the military base, both Native Americans and European settlers farmed some of the area (Kane and Keeton, 1998). Native Americans occupied the region for thousands of years before European settlement and had significant impacts upon some places in the landscape through extensive clearing, burning, and farming,



particularly near to streams. European immigrants settled the area beginning in the early 1800s, and farming was their predominant land use. The U.S. government began acquiring land in 1918 for the infantry school of arms, and the permanent military post was established by Congress in 1920 (Kane and Keeton, 1998). Fort Benning is currently used extensively for U.S. military infantry and tracked vehicle training exercises, yet it retains large areas in semi-natural vegetation.

The land that Fort Benning lies on constitutes part of the Southeastern Mixed Forest Province of the Subtropical Division (Bailey, 1995). This region of high humidity and mild temperatures today is characterized by second growth pine forests of longleaf (*Pinus palustris*), loblolly (*P. taeda*), and slash pines (*P. elliotii*). Frequent, low intensity fires are thought to have been an integral component of the pine forest ecosystem (Glitzenstein et al., 1995). Prior to European settlement, pine forests are thought to have covered much of the landscape, but since then they have been lost or degraded (Frost, 1993; Quicke et al., 1994) mainly as a result of land-use change, timber harvest, and fire suppression (Haywood et al., 1998; Gilliam and Platt, 1999).

Environmental management issues at Fort Benning focus on ways to retain or promote pine forest that support the federally endangered red cockaded woodpecker (*Picoides borealis*) and other habitats important to rare species (Addington, 2004), as well as environmental effects of sediment flux (largely coming from bare areas) (Fehmi et al., 2004). The resource managers are looking for ways to better monitor effects of their efforts to protect habitats of interest and to document impacts of military use on the environment (Addington, 2004). This study focused on broad-scale indicators of landscape change to support monitoring and trend assessment.



### 3. Materials and methods

This study used data from the early nineteenth century and the last quarter of the twentieth century to determine land-cover changes that have occurred in the Fort Benning area since European settlement began. The nineteenth century data are based on land surveys, and the twentieth century data are based on satellite imagery. Although the data sets from each time periods were collected in different ways, we strived to develop procedures that allowed us to compare these data.

#### *3.1 Early nineteenth century data*

To establish a baseline for the study, our analysis relied on witness tree data from land surveys conducted in 1827 and the 1830s. The historic land-survey data were obtained in the form of surveyors' notes and maps from the Georgia Department of Archives and History in Atlanta, from the Alabama Department of Archives and History in Montgomery, and from the Bureau of Land Management in Springfield, Virginia. Because the method of data preparation and integration can affect the results (Mladenoff et al., 2002; Petit and Lambin, 2002,), we describe our techniques in detail as well as relate the origin of the data.

The historical land survey maps were created in the 1800s, when the U.S. government surveyed the country for the purpose of subdividing and selling land. The surveyors partitioned federal lands into square townships that measured six miles on each side (1553 ha). Townships were further subdivided into sections of one square mile (258



ha). Surveyors marked the corners of each section and defined section boundaries by recording witness trees that were in close proximity to the boundary markers. In the 1830s, when the federal government conducted land surveys in the Alabama portion of the area that would become Fort Benning, corner and witness trees were recorded (Foster, 2001; Black et al., 2002). Earlier, when the Georgia portion was surveyed in 1827, the land was divided into roughly equal lots within districts. Districts measured approximately 2330 ha, and lots were 82 ha. The lots were then issued by a lottery. As a part of the land distribution, the Surveyor General surveyed the land, noting the location and species of the corner tree and four witness trees at each corner that marked the boundary of each lot. Unlike in other surveys, the surveyor did not routinely include information regarding bearing or distance from corner, diameter at breast height (DBH), or other indications of tree age or size. Nor did the surveyor include notes describing landscape or understory vegetation.

The land surveys as a whole show great variability in the information that was recorded about witness trees, largely as a result of personal biases of the individual surveyors and differences in their survey methods. Some surveyors recorded only common tree names (which varied greatly), and some reported trees only to the genus level. Others held biases in the species and sizes of trees selected as witness trees. It is often assumed that surveyors were biased towards longer living or larger trees when selecting witness trees (Black et al., 2002). In addition, some surveyors may have exaggerated the amounts of valuable timber species so that land values would be elevated. In spite of these problems, it is largely agreed that “witness tree data is the



largest, most systematic, and most accurate form of data available for the pre-European settlement forests” (Bourdo, 1956; Whitney, 1994; Black et al., 2002).

The historical land survey maps and field notes were used to create a digital Geographic Information Systems (GIS) model of the forests covering the Fort Benning area. The model represents the forests in a Native American agricultural environment prior to extensive European settlement. As such, the model provides baseline conditions for the area currently occupied by the installation. The data extraction process involved georeferencing the historical survey maps, digitizing the location of the trees from the maps, and extracting the attribute species from the maps and survey notes. The historic maps for Georgia were georeferenced to modern U.S. Geological Survey (USGS) maps by using the Fort Benning Final Project and Acquisition Map EM 405-1-2-00 (U.S. Army Corps of Engineers, 1948) because it contained the survey boundaries from the historic maps. Modern aerial photos were occasionally used. The maps were digitized with a GTCO Accutab<sup>TM</sup> 24" x 36" (+/- 0.005" certified absolute accuracy). Data feature points were digitized as points in vector format. Trees were represented as points on the survey maps and in the digital GIS layer. The digitizing and GIS analysis were performed in Arcview 3.2a<sup>TM</sup> and Arc/Info 7.21<sup>TM</sup> (ESRI, Redlands, CA).

Common names of trees from the historic survey maps were assigned scientific names based upon Godfrey (1988). When ambiguous species were encountered, physiographic and habitat associations were used to clarify species (Black et al., 2002). Pine (*Pinus*) was recorded only at the genus level. Many of the corners are labeled only as “stake” or “post” on the plat maps to indicate how the lot corners were marked by the surveyor. In many instances, the species of the post is noted. After consultation with



Mark Cowell (Department of Geography and Geology, Indiana State University, Terre Haute, IN; personal communication; June 1, 1999), who analyzed similar witness tree documents in the Oconee River Valley in central Georgia, we interpreted these posts as trees selected from the immediate vicinity of the corners and therefore indicative of the species that were present at the time of the surveys.

Researchers often directly plot data points representing witness tree locations to describe presettlement vegetation (e.g., Hong et al., 2000). Such point coverage, however, does not adequately describe forest cover, which is continuous. To create a continuous vegetative surface, grids or polygon maps must be extrapolated from the survey points (White and Mladenoff, 1994; He and Ventura, 1995; Brown, 1998a,b; Hong et al., 2000).

A methodology was developed to create the forest cover map by means of Arc/Info 7.21<sup>TM</sup>, GRID<sup>TM</sup>, and ArcView 3.2<sup>TM</sup>. The survey points were first buffered with a radius of 160 m to ensure that each point represented both the corner marker and four witness trees located near most section corners (although a few such points represented as many as eleven trees). We recognize that this pixel size is quite coarse in relation to the satellite imagery. The resulting buffer polygons were then spatially joined to the survey points. Unique identifiers (poly-IDs) were assigned to each buffer polygon, and the point attribute table was edited to include the identifier of the polygon within which it was located. The descriptive information carried in the polygon attribute table was then reformatted to reflect tree type occurrence within the polygon (or survey point cluster). Tree species were assigned a value of either pine, deciduous, or other, with “other” representing small trees, shrubs, and ground cover. On the basis of the ratio of



tree types within the cluster, individual polygons were categorized as pines, deciduous trees, mixed tree species (pine and deciduous), or other.

A raster grid was derived from the polygon coverage through the use of ARC INFO GRID. A grid cell size of 60 m was chosen to match the resolution of the other data in the study. A moving window capturing the neighborhood majority value was applied iteratively to the grid to create a continuous surface of vegetation types for Fort Benning. The final step was to overlay non-forest/cleared areas, which represent large Native American settlements as estimated from archaeological evidence (Foster et al., 2004). Because the locations of smaller settlements were not known, the amount of non-forested land is underestimated on the map.

### *3.2 Twentieth century data*

To document more recent changes in land cover in the Fort Benning area, our study relied on remotely sensed data and created a series of four land-cover maps dating back to the 1970s. Two Landsat 7 Enhanced Thematic Mapper (ETM) images (i.e., path 019/row 037 and path 019/row 038) dated July 24, 1999, were acquired from the Environmental Characterization and Monitoring Initiative (ECMI) Data Repository (<http://sempdata.wes.army.mil>). The ETM images were used for making a current land-cover map of Fort Benning. The data had already been projected and mosaicked by means of nearest neighbor resampling.

North American Landscape Characterization (NALC) data, derived from Landsat Multispectral Scanner (MSS) imagery, were also used in this analysis (Lunetta and Sturdevant, 1993). Landsat MSS has a nominal resolution of 79 m, so we resampled the



NALC data to a 60-m resolution. The NALC data set covering the Fort Benning area is composed of triplicates dated 1974, 1983/86, and 1991 for two scenes (i.e., path 019/row 037 and path 019/row 038). The two scenes for each time period had to be mosaicked in IMAGINE™ before the classification process could begin. The two scenes comprising the mosaic for the 1980s were made in different years; however, given the nature of the landscape and method of comparison used, this time interval was considered acceptable, and the date of mosaic will be referred to as “1983.”

The four data sets (for 1974, 1983, 1991, and 1999) described in the previous paragraphs were then classified to create land-cover maps. Unsupervised classification, which creates a user-defined number of classes based upon spectral response, was used in-house to create 45 spectral classes from the imagery. We combined these 45 classes into six land-cover classes, using a 0.5-m resolution digital color orthophoto (1999) (Obtained from the web site for the wider project: <http://sempdata.wes.army/>) and Land Condition Trend Analysis (LCTA) point data (1991) (Diersing et al., 1992; Jones and Davo, 1997) as reference data. The LCTA protocol uses a modified point intercept method to quantify vegetation cover, ground cover, and disturbance at 1-m intervals on a 100-m line transect with transects randomly placed within a stratified random sampling scheme within an installation. Density and size distribution of woody vegetation are quantified in 600 m<sup>2</sup> plots that are aligned with each transect. The LCTA data tend to underestimate species richness (Prosser et al., 2003) yet provide consistent measures of changes in disturbance, canopy cover and bare ground (Anderson, 2002). In our analysis, the six land-cover classes are water, pine forest, mixed forest (deciduous and pine, areas of sparse forest cover, or areas of transition between forest and non-forest), deciduous



forest, barren or developed land, and non-forest (areas cleared of forest vegetation but with some ground cover that may be grass or transitional areas). These classes are not only distinguishable in the maps; they also reflect ecological conditions of importance to resource managers. To remove any confusion between vegetation and water classes, we created and applied a water mask, using coincident pixels from the classified imagery and the coverage of lakes and major streams. A combination of ARC INFO 7.2.1™, GRID™, ArcView 3.2™, and ERDAS IMAGINE 8.2™ software allowed us to derive land cover from the satellite imagery.

Clouds were not an issue with the MSS data since USGS picks the triplicate data dates upon getting the best, cloud-free data possible whereas the 1999 Landsat 7 Enhanced Thematic Mapper (ETM) images were selected by the Fort Benning resource managers. Therefore we focused on the 1999 image in considering ways to eliminate the presence of clouds and shadows, which were erroneously classified as barren/developed, the cloud and shadow areas were digitized and then overlaid with the 1999 orthophoto. New shapes that more accurately reflected the nature of the vegetation in these cloud-affected areas were digitized on-screen. These polygons were coded with a vegetation type according to interpretations drawn from the orthophoto. The resulting shapefile was plotted with the classified image and adjusted to maintain continuity and blend with neighboring pixels. A grid was created from the cloud-affected-area shapefile, and the classified land-cover image was also converted into a grid. Values from the cloud/shadow grid were used to mask the cloud-affected areas of the classified grid and create an improved land-cover map.



Because all maps are highly generalized representations of reality and contain some error (e.g., Foody, 2002; Brown et al., 1999; Dicks and Lo, 1990; Maling, 1989; Smits et al., 1999), accuracy assessments were conducted to determine how accurately our classification portrayed land cover. We note that accuracy statements that accompany maps of large areas may be erroneous, vague, or nonsite-specific (Foody, 2002). Data, therefore, should be reviewed carefully in the context of its intended use (Fosnight and Greenlee, 2000). LCTA data provided some ground truth information to evaluate the forest type classes, but a preliminary comparison using the LCTA data proved to be variable due to inconsistencies in scale, definitions of cover types, and the sampling dates of the field data. An accuracy assessment was conducted for the 1999 land-cover classification using a 1999 0.5-m digital color orthophoto. The orthophoto was deemed an appropriate source of reference data in the absence of ground truth data. Fifty points per land-cover class were randomly generated and blindly classified on the basis of interpretation of the orthophoto and use of techniques described by Congalton and Green (1999). To facilitate comparison with the other land cover maps, we resampled the 1999 classification to 60-m resolution using nearest neighbor resampling.

Two ways of considering the accuracy of the maps produced from the imagery as compared to the orthophotographs were explored. The omission error quantifies how well cover types identified on the orthophotograph are correctly identified on the map derived from remotely sensing imagery. In contrast, the commission error reveals how well the map depicts cover types on the ground (or in this case, the orthophotographs). Both perspectives are needed to understand the validity of maps.



The ability of the classification to differentiate among forest classes varied within the time series. However, the mixed forest class is the most reliable classification for the 1827 period since the presence of some deciduous and coniferous trees can be overestimated from the survey data. Thus, differences in the area of the “mixed forest” class may be more indicative of changes in the collection (sensors) and methods of processing of the satellite data than of actual changes in forest composition. The 1991 image was classified first and then used as a “reference base” for the earlier images because appropriate reference data were not readily available for dates earlier than 1990. Similarly, no adequate data were available to perform a rigorous accuracy assessment on these earlier classifications. Therefore, some uncertainty is associated with the data. The goal of the project was to evaluate landscape indicators. The quality of the classification is open to improvement if more resources become available or if additional ancillary data is used in the analysis (Klema, 2001). Other papers examining the relationship between accuracy and indices focus on indicators of vegetation cover [e.g., NDVI (Liu et al., 2004)] or particular land-use classes as indicators of landscape condition (e.g., Berlanga-Robles and Ruiz-Luna, 2002).

### *3.3 Landscape indicators*

Indicators of fragmentation were examined to consider ways they can improve understanding of land-cover changes at the landscape level. Landscape metrics were calculated through the use of two computer programs, FRAGSTAT 3.1 (McGarigal and Marks, 1995) and the Analytical Tools Interface for Landscape Assessments (ATtILA) (Ebert et al., 2001). The analysis concentrated on those indicators that differentiated



between broad-scale patterns and revealed independent information (Tischendorf, 2001). ATtILA provides a relatively new suite of tools and is available as an ArcView<sup>TM</sup> extension. It requires Spatial Analyst<sup>TM</sup> and computes landscape metrics based on a land-use/land-cover grid, elevation and slope grids, a streams line theme, a roads line theme, a population polygon theme (county, track, or block), and a precipitation grid. The following ATtILA metrics were of particular interest to this project: land-cover proportions by reporting unit, the amount of non-forested land encountered on steep slopes, diversity metrics, and forest patch metrics.

FRAGSTATS (McGarigal and Marks, 1995), however, was used to compute numerous other landscape metrics. Those reported here include total edge (m); number of patches; an index for the largest patch; mean, range, and coefficient of variation (CV) for the patch area; mean patch area ratio, and the CV of the Euclidean nearest neighbor distance, shape range, and clumpiness. These metrics were selected because they represented statistically significant changes over time in land cover for Fort Benning (Olsen et al., in review). The distance between patches was examined, for it can be important for species that move among a single patch type. The shape index corrects for the size problem of the perimeter-to-area ratio and thus is a measure of overall shape complexity. Clumpiness indicates the proportion of adjacent cells have identical land cover types. Output from each software package was compared and summarized. ATtILA and FRAGSTATS produced the same information describing patch dynamics and landscape composition; therefore, only ATtILA output is presented for the metrics.

The ATtILA and FRAGSTATS packages each offered some advantages. While there is some replication of metrics, FRAGSTATS calculates some metrics that are not



standard output of ATtILA and can output information for specific cover types in addition to summary information for the entire landscape. A benefit of using the ATtILA program is that it allows the calculation and summation of metrics by reporting unit (e.g., catchments or training compartments). Although metrics for the entire installation are useful, metrics for subareas, such as the training compartment, are also effective descriptors, especially as certain military activities are designated to occur only in specific training compartments. Characterizing ecological impacts by compartments allows range managers to adjust training regimes to reduce future ecological effects.

Results were calculated with the use of the default parameters of ATtILA and FRAGSTATS (e.g., 1 pixel minimum patch size, 0 pixel separation of patches, 10 pixel search radius, and 9 pixel moving window). Metrics were calculated for the historical (1827) data, the 60-m NALC data (1974, 1983/86, and 1991), and the ETM+ (1999) land-cover classification. As mentioned earlier, the ETM+ classification was resampled to a 60-m resolution for comparison with the NALC and historical data. Because the resolution of the source data differed originally, caution is necessary in interpreting the results of the analysis. In particular, the magnitude of change reflected by these metrics may be somewhat exaggerated because the historical data are coarse. On the other hand, the magnitude of the change may be underestimated as a result of the misclassification of pine pixels into the mixed forest class. Further caution must be exercised because of the uncertainty associated with the “mixed forest” class. Because they have a distinctive signature, bare areas are rarely misclassified.

Total edge was computed from slightly different data than the other metrics discussed here. While the others were computed from land-cover data that had been



clipped to the boundary of Fort Benning, the edge metrics were calculated from land-cover data that extended beyond the boundary of the installation. This second approach allowed us to calculate the true edge of the class patches instead of accepting the edge as summarily defined by the legal boundaries of the installation.

## **4. Results**

### *4.1 Accuracy assessment*

Overall accuracy for the 30-m classification was 85.6% and for the 60-m resolution classification was 75.20% (Table 1). The commission error was lower than the omission error for all land-cover types except mixed forest. This difference may be because the remote sensing product overestimated the area of mixed forest. The classification of forest classes was expected to contain errors of omission (errors of exclusion) in the pine and deciduous classes and of commission (errors of inclusion) in the mixed classes. Commission occurs when a class on the map includes pixels that should belong to another class (Congalton and Green, 1999). Specifically, the “mixed forest” class may contain pixels of deciduous forest and pine forest, resulting in an underestimation of these classes and an overestimation of mixed forest.

### *4.2 Changes over time in dominant land cover*

Pine forests dominated the landscape at Fort Benning in 1827 (Fig. 2). At that time over 75% of the land area was in pine forest with the second highest category, mixed forest, covering only about 12% (Fig. 3A). While these data are extremely



generalized, the dominance of pine on the historical landscape is supported by the fact that over 90% of the soils at Fort Benning can support longleaf pine (Dale et al., 2002). Furthermore, archaeological evidence suggests that Native Americans almost never cleared forests except adjacent to their settlements (Silver, 1990; Foster et al., 2004).

Dramatic changes in land-cover type occurred in the 137-year period between the 1827 and the 1974 land-cover estimates. By the early 1970s, pine forests had declined to about 25% cover, and deciduous forests dominated the landscape (Figs. 3a and 4). Mixed forest slightly increased in cover. Areas with non-forest vegetation increased to about 15% of cover.

The four-part time series of contemporary land-cover maps from 1974 to 1999 shows persistent landscape features as well as changes in land-cover composition over time (Figs. 4 and 5). Changes in areas of deciduous forest are more commonly associated with riparian areas (Fig. 3b). Areas of bare ground have been fairly stable throughout the period of interest. In recent decades, Fort Benning has experienced a gradual decrease in forest populations along with an increase in non-forest vegetation (Fig. 3a).

The prevailing trend for recent decades is an increase in pine forest and a decline in deciduous forest area (Fig. 3a). The trend may result from ongoing forest management practices at Fort Benning that are aimed at establishing more areas of longleaf pine forest that can support the federally listed red-cockaded woodpecker. These practices include planting longleaf pine seeds and seedlings, as well as regularly setting controlled ground fires, which eliminate hardwoods and other pines while allowing the grass stage of longleaf pine saplings to survive.



The proportion of bare areas has remained relatively stable between 1974 and 1999. Mechanized military training at Fort Benning causes areas to become bare and remain barren of vegetation. Because tracked vehicles are especially detrimental to vegetation, training with tracked vehicles is restricted to certain sacrifice areas. The concept is that it is better for the overall environment to sacrifice a few areas while maintaining high quality ecological conditions in other areas. Hence at Fort Benning the locations of sacrifice areas have not moved in recent decades.

The trends in riparian areas are similar to those reported for the entire installation (Fig. 3b). Land-cover composition within a 1-pixel buffer zone of streams (60 m on each side of stream) was calculated with ATtILA. Notably, the percentage of pine forest in riparian zones appears to have increased between 1974 and 1999, with the rate of change possibly increasing in the 1990s. The percentage of deciduous forest near the streams has remained relatively constant, with the exception of the base wide peak in 1974. Deciduous forests associated with riparian corridors at Fort Benning, therefore, appear to be declining with regard to land-cover composition. However, the amount of bare land found in riparian areas has persisted. The proportion of bare ground and non forest vegetation are particularly important, for they affect stream flashiness, streambed instability, organic matter storage, and stream water dissolved organic carbon concentration (Maloney et al., in press). Furthermore characteristic of these land cover types relate to military training practices at Fort Benning. Although riparian corridors are generally protected from direct training activities, military training within a catchment typically affects the proportion and shape of bare ground and non forest vegetation patches. Furthermore, when the riparian buffer was extended to include 60 m on each



side of the stream (Fig. 3b), base-wide trends in land-cover composition were observed (as depicted in Fig. 3a). We focused on the riparian buffer area because it contains species and supports processes critical to the ecological health of the Fort Benning ecosystem. Resource managers are assessing the ability of indicators within riparian buffers to reflect the overall status of the catchment (Maloney et al., in press). Our analysis suggests that landscape metrics measured within riparian buffers can also reveal information about the status of the system.

#### *4.3 Landscape metrics by land-cover type*

Numerous landscape metrics were computed for each land-cover class through the use of FRAGSTATS (Table 2). We considered the metrics by each land cover type since each type reveals different information about resource management at Fort Benning. After a big increase from 1827 to 1974, the total amount of edge associated with bare/developed patches remained fairly stable throughout the recent decades. This result parallels other patch metrics for the bare/developed class, suggesting that overall impacts of training on bare ground have not changed much in recent years. However, the patch area coefficients of variation have increased, revealing more variability in size of the patches with time. Yet, the amount of edge associated with non-forest vegetation has been increasing probably because more such patches existed in 1999 as compared to 1974. The forest classes also experienced dramatic changes. The amount of edge associated with the pine forest class increased over the entire time period. The amount of edge associated with mixed forest class increased rather sharply between 1827 and 1974 but has been decreasing since 1983. The edge associated with deciduous forest has



vacillated considerably: it increased sharply between 1827 and 1974, decreased abruptly between 1974 and 1983, then increased between 1983 and 1991, and decreased between 1991 and 1999. These abrupt changes in direction of the mixed and deciduous forest types may reflect management actions to promote longleaf pine; such actions have the effect of adding land to the pine forest class and removing land from the other two forest classes, thus changing their distribution on the landscape.

Edge metrics are more meaningful, however, when interpreted in conjunction with other metrics, such as mean patch area. For example, while the number of pine forest patches and total edge has been increasing since 1991, mean patch area for pine patches has remained stable during the same period. Total edge can be important for species that occupy ecotones, the margins of ecological systems (e.g., forest fringes). Patch area is critical for species that require a minimum home range size.

The number of patches associated with each land-cover class increased dramatically between 1827 and 1974 but thereafter fluctuated for bare areas and deciduous and mixed forest and continued to increase for pine and nonforested areas (Table 2). Decreases in edge associated with the bare areas and deciduous and mixed forest classes parallel decreases in the number of patches associated with the forest classes between 1991 and 1999. The larger change over the 137-year period from 1827 to 1974 partially reflects the different mapping technologies but also conveys how significantly human activities have affected the landscape. There was a recent upsurge in the number of bare and developed patches.

The largest patch index (LPI) represents the percentage of the landscape that contains the largest patch of each class and thus the dominance of a single large patch.



The low values of this index (except for the high 73 value for pine forest in 1827) reflect the lack of dominance of any land-cover type (except when pine forests were common before European settlement). These large patches are important for organisms that require large habitats.

Mean patch area and patch area range closely follow the trends of the LPI, already described. Both metrics were highest for pine forests in 1827. All patches were larger in 1827 than in the late twentieth century. The many small patches in all categories in recent decades also result in a smaller patch range. Mean patch area was not highly variable through recent times or among classes because of the presence of many single pixel patches. This stable trend, together with a decrease in the number of patches that are not in pine forest or bare/developed lands and an increase in pine forest composition, indicated that areas of pine forest were getting larger and more continuous. The coefficient of variation of the mean patch size for the bare/developed land has been gradually increasing since 1974. The other land-cover classes vary in this metric for recent times.

The mean perimeter-to-area ratio of the patches did not vary much between classes after 1974. The values were similar for pine, deciduous forest, bare/developed, and non-forest vegetation, while values for mixed forest were consistently higher than the rest. These similarities can be explained by the influence of single-pixel patches (small patches) upon this metric. Mixed forest is associated with the highest values because, as a transitional class, it is most often represented as single pixel or small patch. Deciduous forest is mainly found in riparian areas in large, contiguous patches. Bare/developed and non-forest patches are often the result of some type of managed land use and tend to



occur in relatively simple and confined shapes. Overall, there was not much variability within patch level metrics.

The coefficient of variation (CV) of the Euclidean nearest neighbor distance accounts for variability in measures of the edge-to-edge distance (m) between patches of the same class. There was little within-class variation between 1974 and 1999.

Bare/developed areas were the most widely spread patch type on the landscape followed by non-forest patches. These classes were largely associated with managed land use and comprised a smaller percentage of total landscape composition and number of patches. The forest class values were lower, indicating less distance between forest patches, which were also more prevalent on the landscape at Fort Benning.

Patch shape index-range increased between the 1827 map and recent maps for all classes except pine forest; this increase is a measure of the greater complexity in the shape of areas in land-cover types other than pine forest. After 1974, the shape range index was consistent for the bare/developed areas, but it had a peak in 1983 for the deciduous, mixed, and pine forests. At that same time the non-forest vegetated areas had a dip in their shape range index. These inflection points suggest that the land pattern was most complex in 1983.

The clumpiness index is a measure of patch aggregation. A value approaching 1 indicates a greater degree of patch clumping; zero signifies a random distribution of patches, and -1 indicates disaggregated patches. All land-cover types were more aggregated in 1987 than in other maps for recent decades. The data included negative values for mixed forest from 1974 to 1991, for deciduous forests in 1974, and for pine forests in 1999. Thus the degree of aggregation has fluctuated over time.



#### *4.4 Landscape metrics for entire area of Fort Benning*

Landscape metrics can be useful indicators of overall landscape fragmentation and diversity. Most landscape level metrics changed considerably between 1827 and recent decades (Table 3). Total edge increased more than seven fold between 1827 and 1974. The number of patches and patch density also increased over this 147-year period, and the largest patch index inclined dramatically. In other words a few, large patches occurred in the landscape in 1827, as is evident in a visual examination of the map.

Less change occurred from 1974 to 1999. As with the shape range index, there was an inflection point in 1983 for the landscape metrics for the entire installation. Compared with the other maps for recent decades, the 1983 map had fewer and larger patches with less edge. The amount of edge increased again after 1983 as the largest patch index declined. However, the trends in edge are not consistent with changes in the total number of landscape patches or patch density during the same time period. In general, one might expect the number of patches to increase with total edge, however, the trends do not necessarily agree. Clearly, this indicates a complex landscape.

Furthermore, the patch-level metrics (Table 3) show that increases in the number of patches associated with certain land-cover classes are tempered by decreases in the number of patches associated with other land-cover classes throughout this time period (Table 2), resulting in a fairly stable number of patches at the landscape level. This change demonstrates the necessity of examining the patch-level metrics to get a complete picture of landscape change.



## 5. Discussion

Determining the accuracy of the current map was a necessary step in assigning appropriate confidence in the map products. The process of accuracy assessment highlighted the mixed forest, which may be overestimated. Accuracy assessment is also a useful communication tool in conveying the information revealed by the landscape metrics to decision makers, for it quantifies the classification inaccuracies. It opens the door for mentioning that map accuracy can vary spatially and may be related to land-cover type, terrain, landscape complexity, and land-use patterns (Steele et al., 1998).

Dramatic changes to the Fort Benning landscape occurred between 1827 and 1974. The analyses in this paper reveal how the landscape, once dominated by almost uninterrupted pine forest, transitioned into a mosaic of many different land uses and cover types such as those that occur elsewhere in the contemporary southeastern United State. (Frost, 1993). Even so, the Fort Benning installation still supports more pine forests (more than 35% of the area, Fig. 3A) than most places in the southeast (Frost, 1993). These changes in pine distribution largely explain the current distribution of red-cockaded woodpecker. When pine forest were abundant in the Fort Benning region, the red cockaded woodpecker, which nests in living pine trees, were abundant. However, the expansion of agriculture in the southeastern US in the mid to late 1800s and early 1900s reduced the distribution of pine forests (Kane and Keeton 1998), which greatly contributed to the demise of red cockaded woodpeckers. Yet, the situation at Fort Benning was different. These maps reveal the success of management programs at Fort Benning in expanding the distribution of pine forests in recent decades. The broad



distribution of so many longleaf pine trees is a key reason that Fort Benning is a major component of the Fish and Wildlife Service's management plan for red cockaded woodpecker. Characteristics of the mixed forest are important to monitor, for they reveal information about areas that could be managed to support pine forests.

The land-cover map created from the 1827 witness tree data is critical to understanding historical conditions on the land. Yet the wide gap between the 1827 map and the satellite imagery of the 1990s causes some problems in interpretation of the trends. Therefore, we are exploring whether historical aerial imagery can be used to construct land-cover maps for some of the intervening decades.

The map of nineteenth century land cover produced from the witness tree data is useful even though the data do not provide continuous coverage and are highly variable. By augmenting the witness tree data with archeological evidence, the 1827 map could include areas of bare ground and sites developed for human use. In particular, the 1827 map shows that pine forest dominated the area even though native forests were actively burned as part of land clearing in the central region of Georgia. Similar witness tree data have been used to verify the presence of extensive pine stands in frequently burned areas of the Missouri Ozarks before European settlement (Batek et al., 1999) and in northern Florida (Schwartz, 1994).

Subtle changes to land cover at Fort Benning occurred in recent decades. The area of pine forest has been gradually increasing in the past 25 years (Fig. 3A). Improved monitoring techniques coupled with an aggressive management strategy for perpetuating pine forest at Fort Benning (Waring et al., 1990) have likely been the cause for the increase in pine and the corresponding decrease in deciduous forest. This management



strategy includes harvesting timber from stands other than longleaf pine and thinning and burning areas of longleaf pine (Haywood et al., 1998; Gilliam and Platt, 1999; Provencher et al. 2001).

While the percentage of non-forest vegetated land has slightly increased, the metrics show that number of non-forest vegetated patches has increased tremendously in the past 25 years. Consequently, the size of these patches has decreased. Altered management has likely constrained certain land uses to smaller geographic areas.

The selection of methods and landscape metrics for analysis of land cover changes was a key focus of this analysis. Landscape metrics are essential to quantifying changes in land cover. Of the large set of metrics available in the FRAGSTATS and ATtILA software, we found the most useful metrics for depicting changes in land cover at Fort Benning to be percent cover, total edge with border, number of patches, mean patch area, patch area range, the coefficient of variation in patch area, the perimeter-to-area ratio, Euclidean nearest neighbor distance, shape range, and clumpiness. While these last two metrics show little change over time, they distinguished between land-cover categories quite well. In terms of the entire landscape of Fort Benning, the metric for total edge changed most over recent time periods and was useful for analysis.

## **6. Conclusions**

Data collected for disparate purposes can be used to help develop an understanding of land-cover changes over time and are often necessary to further our knowledge of historic conditions on a given landscape. For the entire Fort Benning



landscape, the values of metrics for 1827 were very different from the values for recent decades. While the changes between 1827 and 1974 may be somewhat exaggerated due to data constraints, we can conclude that the nineteenth century landscape at Fort Benning was composed largely of uninterrupted pine forest with some deciduous forests found in riparian corridors and some open areas associated with Native American settlements. Land cover and land use in the 1970s were considerably different. Following decades of farming, military training activities had a pronounced affect upon the landscape. Heavy training activities resulted in areas of sparse land cover and bare ground. Interestingly, these areas have largely persisted on the landscape throughout the 1980s and 1990s. This result not only emphasizes the lasting footprint that military activities have on the landscape but also highlights the efforts made by management to confine heavy training exercises to certain sacrifice areas. Another interesting trend occurred in the 1990s. Pine forests have been on the rise as is reflected in both landscape composition and patch dynamics such as largest patch size, number of patches, and total edge. Management efforts at Fort Benning have focused upon managing for longleaf pine. These efforts appear to be decreasing hardwood invasion in favor of pine species in many areas on the installation.

Examining a suite of landscape metrics over time was useful for summarizing, describing, and assessing land-cover change at Fort Benning. The FRAGSTATS and ATtILA programs were relatively simple to use and provided information pertinent to understanding and managing the land. Therefore, we encourage resource managers to use landscape metrics to analyze changes in patterns of land cover over time to examine how human activities have affected an area.



At Fort Benning, these results are being used to understand the broad-scale effects of existing training and forest management practices and to help design new military training areas. In late 2004, construction began on an approximately 730 ha Digital Multipurpose Range Complex (DMPRC) at Fort Benning. To prepare for the DMPRC, forest is being cleared, and soil is being moved. The new area will use new digital technology for training with the Bradley Fighting Vehicle and the Abrams M1A1 tank and support the influx of more soldiers expected at Fort Benning. The alternative action plan that was selected will have negative environmental impacts on endangered species and their habitats (Environmental Impact Statement for Digital Multi-Purpose Range Complex, 2004). The results of this study on landscape metrics and other research on indicators supported by the SERDP Ecosystem Management Program (SEMP) are being used to design and monitor the impacts of the DMPRC (Westbury, 2004). Changes in the area and in patch dynamics of pine forest and bare/undeveloped land are being given prime attention.

Furthermore, work has already begun on obtaining and classifying historical aerial photography from the 1940s and 1950s. The characterization of additional time periods between 1827 and 1974 will be extremely useful in bridging the gap in our understanding of the landscape dynamics between the nineteenth century landscape and the established military base of the 1970s.

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Table 1. Accuracy assessment of classified imagery from 30 m resolution image as compared to orthophotography

| Classified image                                  | Land cover observed from<br>orthophotography |          |           |           |           | Total | <b>Omission (%)<sup>2</sup></b> |
|---|--|----------|-----------|-----------|-----------|-------|---------------------------------|
|   | 1. B/D                                       | 2. NFV/T | 3. DF     | 4. MF     | 5. PF     |       |                                 |
| 1. Bare/developed (B/D)                           | 46   | 4        | 0         | 0         | 0         | 50    | <b>8</b>                        |
| 2 Non forest vegetation /<br>transitional (NFV/T) | 2  | 42       | 1         | 4         | 1         | 50    | <b>16</b>                       |
| 3. Deciduous forest (DF)                          | 0  | 0        | 40        | 8         | 2         | 50    | <b>20</b>                       |
| 4. Mixed forest (MF)                              | 0  | 0        | 2         | 40        | 8         | 50    | <b>20</b>                       |
| 5. Pine forest (Pf)                               | 0  | 0        | 2         | 2         | 46        | 50    | <b>8</b>                        |
| Total   | 48   | 46       | 45        | 54        | 57        | 250   |                                 |
| <b>Comission(%)<sup>1</sup></b>                   | <b>4</b>                                     | <b>9</b> | <b>11</b> | <b>26</b> | <b>19</b> |       |                                 |
| Overall accuracy: 85.6%                           |  |          |           |           |           |       |                                 |
| Kappa statistic: 0.82                             |  |          |           |           |           |       |                                 |
| Producer's Accuracy (%)                           | 96   | 91       | 89        | 74        | 81        |       |                                 |
| User's Accuracy (%)                               | 92   | 84       | 80        | 80        | 92        |       |                                 |
| Kappa statistic                                   | 0.901  | 0.8039   | 0.7561    | 0.7449    | 0.8964    |       |                                 |

<sup>1</sup>Omission error percentages refer to the number of reference points (from orthophotography) omitted and incorrec classified relative to the total number of reference points per cell

<sup>2</sup>Comission error percentages refer to the number of misclassified pixels in the land cover class relative to the total number of classification points per class



Table 2. Metrics by land-cover class and over time at Fort Benning, GA.

| Land cover            | Metric  | Year    |        |        |        |        |
|-----------------------|---|---------|--------|--------|--------|--------|
|                       |   | 1827    | 1974   | 1983   | 1991   | 1999   |
| Bare/developed        | Total edge with border (km)                     | 68      | 610    | 847    | 720    | 989    |
|                       | Number of patches                               | 9       | 2274   | 2905   | 2501   | 3460   |
|                       | Largest patch index                             | 0.679   | 0.0149 | 0.0142 | 0.0149 | 0.0176 |
|                       | Mean patch area (ha)                            | 173.32  | 0.95   | 0.89   | 0.98   | 0.85   |
|                       | Patch area range (ha)                           | 480.24  | 7.56   | 7.2    | 7.56   | 9      |
|                       | Patch area coefficient of variation (CV)        | 94.39   | 86.39  | 94.57  | 104.98 | 108.71 |
|                       | Mean perimeter/area ratio                       | 73.60   | 528.63 | 547.66 | 540.89 | 567.74 |
|                       | CV of Euclidean nearest neighbor distance (NND) | 255.22  | 68.22  | 61.59  | 62.66  | 60.94  |
|                       | Shape range                                     | 1.20    | 1.45   | 1.35   | 1.45   | 1.65   |
|                       | Clumpiness                                      | 0.93    | 0.28   | 0.26   | 0.28   | 0.25   |
| Deciduous forest      | Total edge with border (km)                     | 217     | 6600   | 3914   | 4528   | 4320   |
|                       | Number of patches                               | 47      | 18223  | 10670  | 12601  | 11257  |
|                       | Largest patch index                             | 1.376   | 0.0284 | 0.0209 | 0.0257 | 0.029  |
|                       | Mean patch area (ha)                            | 118.24  | 0.97   | 0.98   | 0.97   | 0.84   |
|                       | Patch area range (ha)                           | 1015.92 | 14.76  | 10.8   | 13.32  | 15.12  |
|                       | Patch area CV                                   | 153.30  | 94.61  | 89.16  | 94.52  | 109.74 |
|                       | Mean perimeter/area ratio                       | 126.38  | 540.00 | 527.84 | 533.47 | 562.48 |
|                       | CV of Euclidean NND                             | 105.53  | 22.35  | 34.54  | 32.86  | 36.24  |
|                       | Shape range                                     | 1.50    | 2.01   | 2.06   | 1.88   | 2.36   |
|                       | Clumpiness                                      | 0.93    | -0.06  | 0.15   | 0.11   | 0.13   |
| Mixed forest          | Total edge with border (km)                     | 307     | 7297   | 8071   | 7197   | 5763   |
|                       | Number of patches                               | 50      | 20494  | 22413  | 20185  | 17027  |
|                       | Largest patch index                             | 1.474   | 0.0108 | 0.0182 | 0.0135 | 0.0162 |
|                       | Mean patch area (ha)                            | 173.30  | 0.59   | 0.72   | 0.60   | 0.68   |
|                       | Patch area range (ha)                           | 1087.92 | 5.4    | 9.36   | 6.84   | 8.28   |
|                       | Patch area CV                                   | 131.98  | 69.28  | 85.47  | 72.12  | 84.85  |
|                       | Mean perimeter/area ratio                       | 96.31   | 598.38 | 575.62 | 598.13 | 584.40 |
|                       | CV of Euclidean NND                             | 64.53   | 24.62  | 21.95  | 25.84  | 28.90  |
|                       | Shape range                                     | 1.53    | 1.67   | 2.27   | 1.56   | 1.57   |
|                       | Clumpiness                                      | 0.93    | -0.18  | -0.20  | -0.17  | 0.01   |
| Non-forest vegetation | Total edge with border (km)                     | 16      | 2527   | 2839   | 3197   | 4095   |
|                       | Number of patches                               | 6       | 8237   | 8950   | 9859   | 12325  |
|                       | Largest patch index                             | 0.159   | 0.0317 | 0.0203 | 0.0331 | 0.0304 |
|                       | Mean patch area (ha)                            | 54.24   | 1.01   | 0.97   | 1.04   | 0.92   |
|                       | Patch area range (ha)                           | 113.04  | 16.56  | 10.44  | 17.28  | 15.84  |
|                       | Patch area CV                                   | 64.57   | 104.01 | 98.30  | 109.73 | 119.10 |
|                       | Mean perimeter/area ratio                       | 103.29  | 535.39 | 537.18 | 533.36 | 559.42 |
|                       | CV of Euclidean NND                             | 131.90  | 42.67  | 43.33  | 38.08  | 34.66  |
|                       | Shape range                                     | 0.50    | 2.21   | 1.83   | 2.43   | 2.43   |
|                       | Clumpiness                                      | 0.91    | 0.20   | 0.18   | 0.17   | 0.12   |
| Pine forest           | Total edge with border (km)                     | 371     | 4311   | 5777   | 6108   | 6501   |



|                           |         |        |        |        |        |
|---------------------------|---------|--------|--------|--------|--------|
| Number of patches         | 19      | 11859  | 16021  | 16790  | 17450  |
| Largest patch index       | 73.517  | 0.0581 | 0.0628 | 0.0351 | 0.027  |
| Mean patch area (ha)      | 3038.31 | 1.08   | 0.94   | 0.95   | 1.01   |
| Patch area range (ha)     | 54280.8 | 30.6   | 33.12  | 18.36  | 14.04  |
| Patch area CV             | 397.75  | 108.30 | 124.54 | 108.82 | 108.44 |
| Mean perimeter/area ratio | 290.86  | 525.34 | 554.29 | 547.01 | 541.53 |
| CV of Euclidean NND       | 70.83   | 34.03  | 29.34  | 28.61  | 24.47  |
| Shape range               | 2.86    | 2.83   | 3.10   | 2.17   | 2.21   |
| Clumpiness                | 0.94    | 0.13   | 0.04   | 0.02   | -0.02  |



Table 3. Metrics for entire landscape at Fort Benning, GA

| Year | Total edge with<br>border (km) | Number<br>of patches | Patch<br>density | Largest<br>patch index |
|------|--------------------------------|----------------------|------------------|------------------------|
| 1827 | 6624                           | 131                  | 0.1774           | 73.5166                |
| 1974 | 49405                          | 61312                | 115.0106         | 0.0581                 |
| 1983 | 48498                          | 61256                | 114.9055         | 0.0628                 |
| 1991 | 52672                          | 62254                | 116.7626         | 0.0351                 |
| 1999 | 93360                          | 61837                | 115.9954         | 0.0304                 |



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Fig. 1. Fort Benning, Georgia.

Fig. 2. Forest-cover map of Fort Benning, Georgia, in 1827.

Fig. 3. Land-cover composition at Fort Benning, Georgia, for (A) the entire installation and (B) riparian buffers (60 m wide).

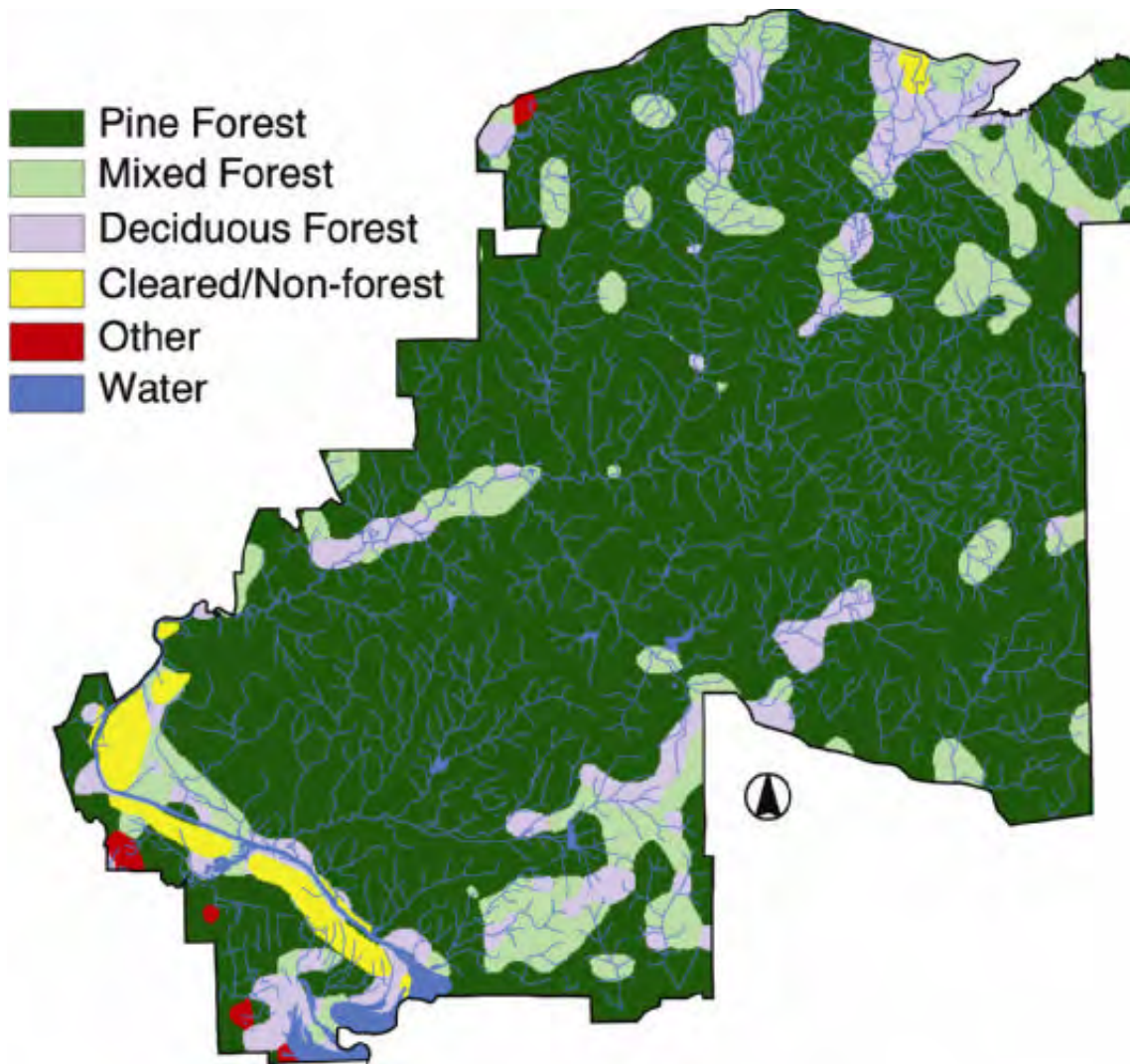
Fig. 4. Land-cover classification for Fort Benning, Georgia, based on ETM+ imagery for 1999.

Fig. 5. Land-cover classification for Fort Benning, Georgia, based on NALC data for (A) 1974, (B) 1983/86, and (C) 1991.



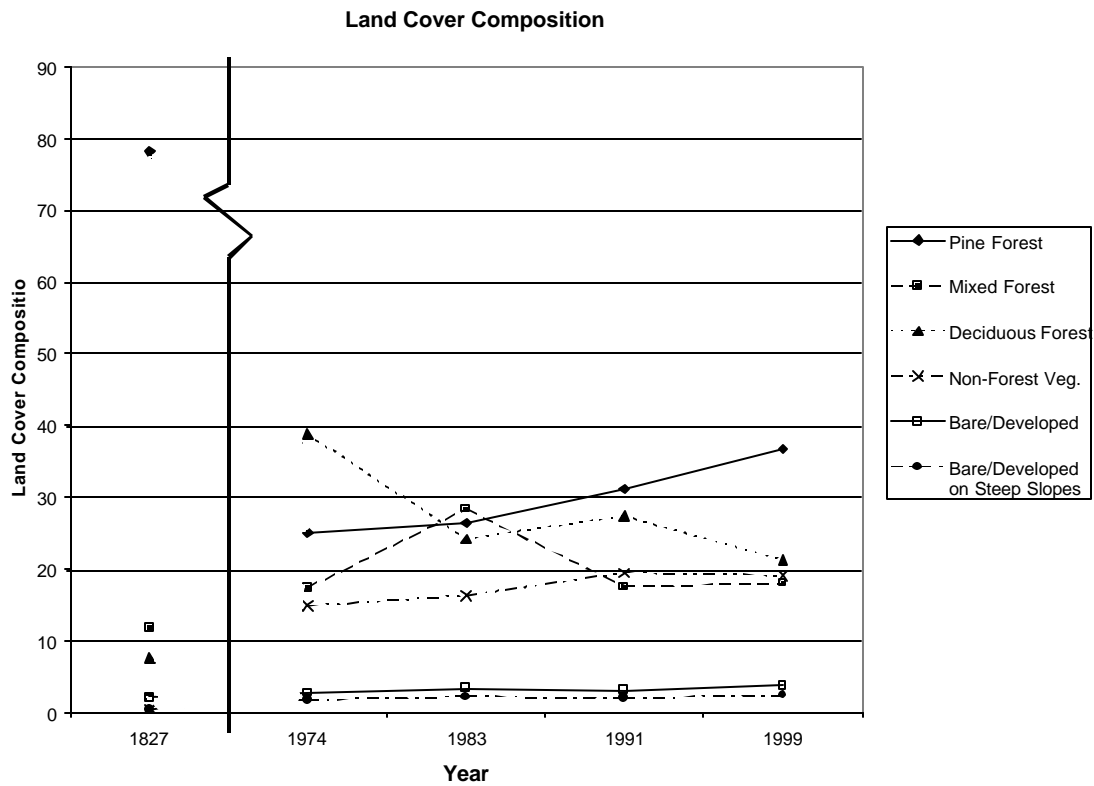
Figure 1



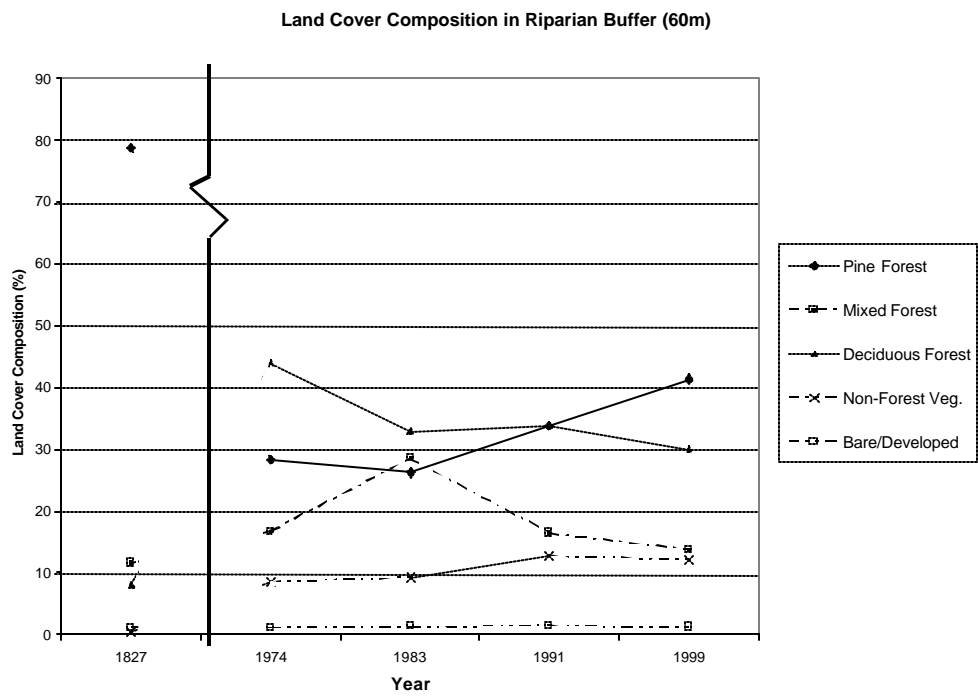




## A) Land Cover Composition – I prefer to add unit (%)?



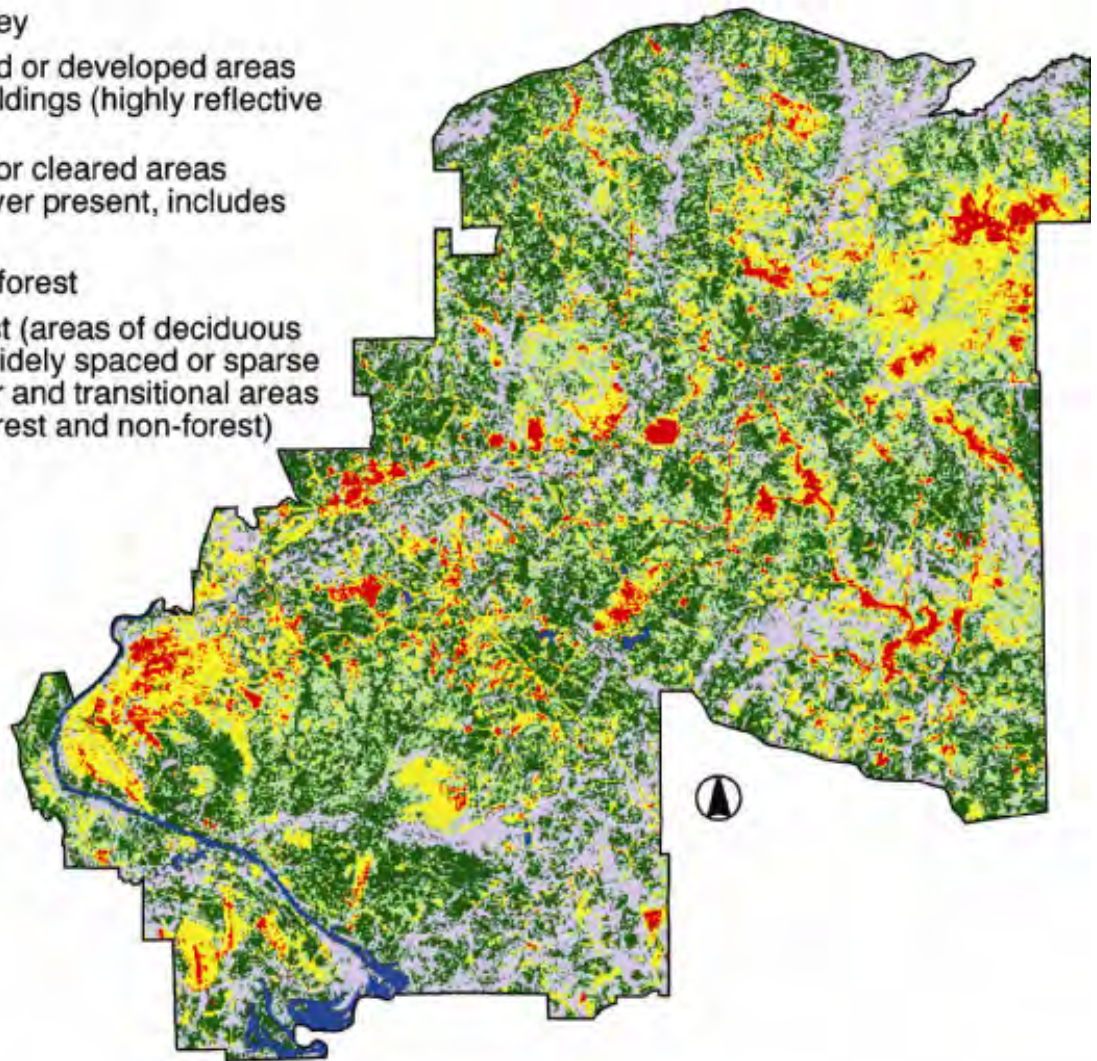
## B)



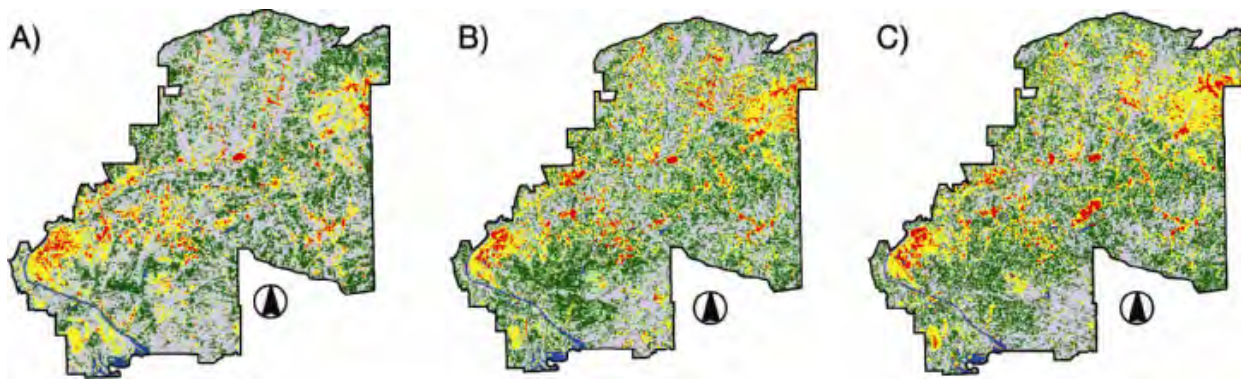


Land-Cover  
Classification Key

- Bare ground or developed areas  
such as buildings (highly reflective  
surfaces)
- Non-forest or cleared areas  
(ground cover present, includes  
lawns)
- Deciduous forest
- Mixed forest (areas of deciduous  
and pine, widely spaced or sparse  
forest cover and transitional areas  
between forest and non-forest)
- Pine forest
- Water









## **Time-Series Analysis over 172 Years Using Landscape Metrics at Fort Benning, Georgia**

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## **Abstract**

Time series of thematic land-cover maps are used to measure changes in land cover over time. However, pixel-to-pixel comparisons of such maps are often not advisable when these maps are generated from different sources (i.e., satellite data, aerial photography, or historical land survey data). The purpose of this study was to examine the historical changes in land cover from 1827 to 1999 using landscape metrics calculated on maps created from different sources. Regression and power law analyses were conducted to identify significant trends and threshold effects. Results indicated that since 1827 the landscape has become more fragmented with the introduction of farming, military training, and forest management practices.



## INTRODUCTION

Determining patterns of change over time on a landscape is necessary for interpreting influences on ecological interactions. *Land use* refers to the management regime humans impose on a site (e.g., pastures or plantations), and *land cover* is a descriptor of the status of the vegetation at a site (e.g., deciduous forest or grassland). The ecological effects of land use/land-cover (LULC) change include changes in biodiversity, habitat availability, and soil erosion and deposition (Dale 1997). These changes can be beneficial to particular endangered species (e.g., prescribed burns to maintain red cockaded woodpecker habitat) or detrimental (e.g., fragmentation of rangeland leading to loss of rare plant habitat). Today, land cover is principally altered by human activity, including agriculture, urbanization and industrialization, mining, forest planting, and harvesting (Dale et al. 2000; Turner et al. 2001). Land-cover patterns are a function of environmental conditions and human use, but how these conditions and uses affect current land cover is still a critical area of ecological research (Wu and Hobbs 2002). Detecting change in LULC is important to informing management decisions on how particular interventions affect the sustainability of natural resources (e.g., Weiers et al. 2004).

LULC maps can be produced from various sources, including field survey data, topographic maps, aerial photography, and satellite imagery. LULC maps can be produced using the expert knowledge of a cartographer or with the use of pattern-recognition algorithms that aggregate similar spectral characteristics into information classes or categories of digital data (Jensen 2004). To assess the accuracy of a classification, each category in a thematic map is spatially compared to finer-grained data—e.g., field data or high-resolution aerial photography that accurately represents the land cover at that snapshot in time (Congalton and Green 1999).

Historical data sources, such as witness tree records dating back to the 1800s, are necessary to understanding long term patterns of land-cover change (He et al. 2000). Yet these data are quite different in resolution and information from contemporary remotely-sensed data sources such as satellite imagery and aerial photography (He et al. 2000; Manies and Mladenoff 2000; Manies et al. 2001; and Schulte and Mladenoff 2001). A key challenge is determining how best to combine historical information based on field observations with remotely sensed data.

Land-cover maps representing a time series can be used to examine trends. Several methods can be used to compare land-cover maps produced over time. Nonparametric categorical analysis of thematic maps involves the use of cross-tabulation tables to compare pairs of thematic maps. The cross-tabulation tables are analyzed using measures of association such as chi square statistics and the Kappa index of agreement (Congalton and Green 1999). Multiple pairwise comparisons between time series of thematic maps are time consuming. Singh (1989), Eastman and McKendry (1991), and Coppin et al. (2004) have provided guidance on quantitative and qualitative digital-image change detection methods that rely on parametric pixel-to-pixel or pairwise comparisons and nonparametric categorical analyses. Pairwise quantitative analyses include image differencing, ratioing and regression, and change vector analyses (Eastman and McKendry 1991). For example, satellite image differencing is a change detection method whereby the reflectance values or their transforms (e.g., vegetation indices) of two digital images or a series of images, usually from different dates, are subtracted from each other to detect a change—i.e.,  $(\text{Time 1} - \text{Time 2}) + C = \Delta\text{Time}$ , where  $C$  equals a constant to remove negative values (Singh 1989; Eastman and McKendry 1991; Washington-Allen et al. 1998). Eastman and McKendry (1991) introduced a number of time series analyses for multitemporal image comparisons, including image deviation, change vector analysis, time sequencing and profiles, and principal components analysis (PCA). Washington-Allen et al. (2004) recently used cluster analyses of a vegetation index time series to create spatiotemporal maps of plant community declines. However, it is often necessary to compare thematic maps that were created independently using data from different sources, of different resolutions, purposes, and quality (e.g., Jensen et al. 1993, Petit and Lambin 2001). While standardization methods exist, they still may not adequately account for differences in spatial grain, data quality, errors due to misclassification, and errors due to misregistration. Consequently, the uncertainties introduced by these errors do not justify the application of a pixel-to-pixel comparison.



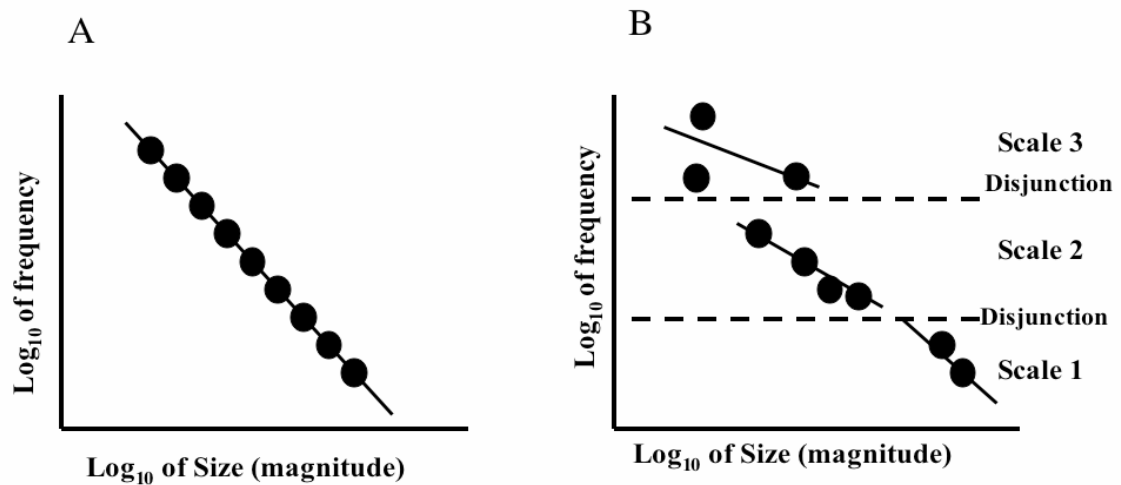
An alternative approach to pairwise comparisons of a time series of LULC maps is the use of landscape metrics (e.g., Spies et al. 1994; Washington-Allen 2003). Landscape metrics were developed to quantify changes in the composition and configuration of landscape elements (O'Neill et al. 1988; Turner et al. 2001). The fundamental unit of a landscape is an element or patch (McGarigal and Marks 1995, Turner et al. 2001). A patch is defined as an area of contiguous pixels of the same land-cover type. Patches are usually subjectively chosen but objectively measured using landscape metrics. For example, more than 55 different metrics have been developed to measure changes in landscape configuration and composition, including the area of classes, diversity, clumping and dispersion, and the number of edges, classes, and patches. Patches of habitat on a landscape characterize the spatial arrangement of conditions that can support particular organisms. As these patches change in shape and extent over time, the landscape becomes more or less suitable for specific organisms. For example, cowbirds (*Molothrus ater*) require large areas of edges between grasslands and forests in order to have adequate conditions for both foraging and nesting. Regression analysis of a time series of landscape metrics can identify significant trends through time.

Previous uses of landscape metrics for change detection have shown that these measures can capture changes such as landscape destruction and rehabilitation (Herzog et al. 2001); vulnerability to wildfires, smoke production, forest pathogens, and insect disturbance (Hessburg et al. 1999); and differences between current and historical conditions (Hessburg et al. 1999).

Characterization of a landscape's patch dynamics can provide information on the organization and stability of landscape composition and configuration. Self-organized criticality (SOC) has been used as an organizing theme to explain landscape composition and configuration (Bak and Tang 1989, Milne 2000). If the processes that form patches are in balance with each other, a frequency plot of patch-size classes will show scale invariance [Fig. 1(A)] (Milne 2000). Criticality means that the balance of processes that result in a particular landscape's patch-size class distributions are poised on a threshold—i.e., show scale invariance. However, when a mismatch of processes has occurred, then disjunctions or departures from linearity are observed, and a threshold or thresholds at multiple scales (the size class of patches) have been exceeded [e.g., Fig. 1(B)] (Ludwig et al. 2000). Milne (2000) has hypothesized that the number of different plant community patch-size classes within a landscape is either a function of different processes acting at different spatial and temporal scales or the consequence of one process acting at multiple scales. Milne also has hypothesized that in regard to ecological data, breaks in the fit line due to significant outliers or large residuals are indicative of non-equilibrium antagonistic processes, or sets of processes affecting landscape patches at different scales (Milne 2000).

For example, tallgrass prairie dynamics are usually constrained by fire, herbivory, and climatic variables (Briggs et al. 1998). However, the balance between these processes is constantly changing, and thus, the tallgrass prairie has been termed a non-equilibrium system (Knapp and Seastedt 1998). Analysis of patch dynamics can also detect thresholds of change. In other words, the dynamics of a self-organized landscape are tightly coupled and yet antagonistic because exogenous and endogenous constraints mutually interact to roughly balance one another (Perry 1995; Li 2000).





When landscape metrics are examined over time, significant trends can be detected, and landscape composition and configuration can be evaluated for threshold effects. The overall goal of the present effort was to create cost-effective tools that will allow resource managers to understand how metrics and time-series analysis can be used to quantify changes in land cover at Fort Benning, Georgia, between 1827 and 1999. Griffith et al. (2003) documented the importance of changes in forests in the southeastern US and showed that recent trends in LULC were best analyzed at an ecoregional scale of resolution. Building off of previous calculations of landscape metrics for the area (Dale et al. in press), here we explore the use of regression analysis to determine if the time series of metrics exhibit significant trends over 172 years. We hypothesize that prior to widespread European settlement in the late 1800s, the landscape was quite different from what it is today. The presettlement landscape would represent baseline conditions of relatively pristine, clumped, and unfragmented patches when compared to contemporary landscapes. Consequently, a presettlement landscape should be detected as an anomaly by landscape metrics in a time series that includes contemporary landscapes.

### STUDY SITE

The Fort Benning army installation occupies 73,503 ha in Chattahoochee, Muscogee, and Marion Counties of southwest Georgia and Russell County of southeast Alabama (Fig. 2). Fort Benning was established in 1920 by Congress on plantations and farmlands south of Columbus, Georgia, for the U.S. military (Kane and Keeton 1998). Fort Benning is the site of the U.S. Army Infantry, Airborne, and Ranger Schools, and the 29th Infantry Regiment, which teaches soldiers to operate the Bradley fighting vehicle. The annual number of troops at the site ranges between 18,000 and 23,000.

The area now occupied by Fort Benning was originally occupied by Native Americans, who built settlements along the rivers. There, they hunted and grew crops before being supplanted by Europeans, who practiced intensive agriculture on the land from the 1800s until the base was established (Kane and Keeton 1998). Probably because much of the installation was protected from land development, it now



supports ecological systems and species that were once common in the southeastern United States but are now quite rare, including the red-cockaded woodpecker (*Picoides borealis*), longleaf pine (*Pinus palustris*), and the gopher tortoise (*Gopherus polyphemus*).

The Fort Benning area has a mean annual temperature of 18.4°C and a mean annual precipitation of 123 cm (Kane and Keeton 1998). The site is bisected by the Fall Line transition zone between the Southern Appalachian Piedmont and Upper Coastal Plain physiographic provinces. Most of the soils are

highly-weathered Ultisols derived from coastal plain or alluvial deposits. Sands and loamy sands are common on upland sites, while sandy loams and sandy clay loams are found in the valleys and riparian zones.

Today, the base is characterized by approximately 36,000 ha of secondary-growth pine and mixed pine/hardwood forests, including upper-canopy dominants longleaf, loblolly (*P. taeda*), slash (*P. elliotii*), and sand pines (*P. clausa*) and various oak (*Quercus*) species at midstory. The understory is dominated by mixed grasses and forbs, including the native pyrophytic grasses, sandhill dropseed (*Sporobolus junceus*) and arrowfeather three-awn (*Aristida purpurascens*). Since the 1960s, much of the Fort Benning installation has been managed to promote longleaf pine, which involves low-level fire every 1 to 5 years (Provencher et al. 2001).

## **METHODS**

### **Data Preparation**

Land-cover maps from 1827, 1974, 1983, 1991, and 1999 were used in this paper to explore how the landscape has changed at Fort Benning over time (Table 1). Data preparation and processing have been previously described by Olsen et al. (2001) and Dale et al. (in press). The 1827 map was created from historical land survey maps and field notes of the forests covering the Fort Benning area from the early 1800s (Foster 2001; Olsen et al. 2001). This historical map is a representation of the forests in a Native American agricultural environment prior to extensive European settlement.

Land-cover maps representing the 1970s, 1980s, and early 1990s were generated from 79-m resolution Landsat Multispectral Scanner (MSS) satellite images that compose the North American Land Characterization (NALC) triplicate dataset (Lunetta and Sturdevant 1993). The MSS data had been resampled from 79-m resolution to 60-m for use in the NALC program (Lunetta and Sturdevant 1993, Table 1). The 1999 land-cover map of Fort Benning was created from classification of two Landsat 7 Enhanced Thematic Mapper Plus (ETM+) images dated July 24, 1999. The data were acquired from the Environmental Characterization and Monitoring Initiative (ECMI) Data Repository and had already been projected and mosaicked using nearest-neighbor resampling (<http://sempdata.wes.army.mil>). The same unsupervised classification procedure was used on the MSS and ETM+ data. An unsupervised classification procedure was used to create 45 user-defined classes based upon spectral response from the imagery. These 45 classes were then combined into six land-cover classes to match the 1827 land cover map using a 0.5 meter resolution digital color orthophoto (1999) and Land Cover Trend Analysis (LCTA) point data (1991) (Diersing et al. 1992) as reference data. The six classes include: water, pine forest, mixed forest (deciduous and pine, areas of sparse forest cover, or areas of transition between forest and non-forest), deciduous forest, barren or developed land, and non-forest (areas cleared of forest vegetation but with some ground cover that may be grass or transitional areas).

### **Accuracy Assessment**

Errors of omission (errors of exclusion) in the pine and deciduous classes and of commission (errors of inclusion) in the mixed classes are expected. Commission occurs where one class includes pixels that should be classified as something else (Congalton and Green 1999). Specifically, the “mixed forest” class may contain pixels of deciduous forest and pine forest, resulting in an underestimation of these classes



and an overestimation of mixed forest. All maps are representations or models of reality. As such, maps are highly generalized and contain error (Brown et al. 1999; Smits et al. 1999; Foody 2002). Appropriate reference data, including aerial photographs and contemporary field data, were not readily available to conduct an accuracy assessment of the 1974, 1983, and 1992 land cover maps.. Consequently, there were no adequate data available to perform an accuracy assessment on these classifications, and some unquantifiable uncertainty is associated with these data. Most notable are potential misclassifications in the mixed forest class in 1983 and the deciduous forest class in 1974 due to differences in image characteristics and quality. As a result, the output statistics may be skewed and over-represent those classes for those dates. However, an accuracy assessment was conducted to determine if the generated map for 1999 was an appropriate representation. Reference points were randomly generated and blindly classified using digital orthophotos (as described in Dale et al. in press) following techniques described by Congalton and Green (1999). The quality of the classification is open to improvement if more resources become available. An accuracy assessment was conducted for the 1999 land-cover classification using the 1999 0.5-m digital color orthophoto. Overall accuracy for the 30-m classification was 85.6%. Accuracy decreased to 75.20% when the data was resampled to a 60-m resolution. Appropriate reference data were not available to assess the earlier classifications.

### **Calculating the Metrics**

While several landscape metrics describing a diversity of land-use types and how they may affect land-management activities were considered in this analysis, candidate metrics derived from our first analysis (Dale et al. in press) were examined more closely based upon their statistical significance (Table 2). These include, type of land cover, measures of cover type composition, area, density and edge (e.g., the radius of gyration range [gyra, ra]), shape (including the different descriptive statistical estimates of the fractal dimension), isolation and proximity (e.g., similarity dimensions), contagion and interspersion (e.g., the Interspersion/Juxtaposition Index [IJI]), connectivity (e.g., the patch cohesion index), and diversity.

The Analytical Tools Interface for Landscape Assessments (ATtILA) (Ebert et al. 2001) and FRAGSTATS 3.1 (McGarigal and Marks 1995) were used to evaluate the land-cover maps and calculate landscape metrics. While there is some overlap of metrics between these software packages, both FRAGSTATS and ATtILA calculate metrics that are unique to each package, e.g., ATtILA calculates a patch cohesion index (Table 2) that is not present in FRAGSTATS. Additionally, FRAGSTATS outputs information at the class spatial scale that are not available in ATtILA.

Metrics were computed for the 1827 map and the land-cover maps generated from the NALC classifications and the 1999 ETM classification using both ATtILA and FRAGSTATS. The ETM+ classification was resampled to a 60-m resolution for comparison with the NALC and historical data. Output from each software package was summarized and compared. ATtILA and FRAGSTATS revealed much of the same information describing patch dynamics and landscape composition.

To compensate for isolated and/or misclassified pixels, a second set of images was derived from the initial classifications using a 3 X 3 moving window filter to essentially “smooth” the landscape. This filtering approach assigned the majority value to the center pixel in the moving window. Where no majority was found, the original pixel value was retained. This process eliminated most single-pixel patches and resulted in more continuous vegetation types. The filtering process resulted in a minimum mapping unit (mmu) of 160 X 160 m. Metrics were also computed for the filtered/smoothed data, and the two sets of results were compared. Time series analysis showed that trends were similar for filtered and unfiltered data for most classes, although they did vary in magnitude and shape. Consequently, the time-series analysis was conducted using the filtered data which had removed the “noise” of classes below the mmu of 160 X 160 m.



### **Time-Series analysis**

Time series of class-level and landscape-level metrics were examined to test for significance. A significant slope ( $\beta$ ) is a measure of the direction of trend [i.e., stable (0), increasing ( $0 < \beta \leq +1$ ), or decreasing ( $0 > \beta \geq -1$ )]; and the magnitude of the coefficient of determination ( $r^2$ ) from a linear or polynomial regression is a measure of a landscape metrics trend (Yafee and McGhee 2000). Following accepted methodologies for environmental data, significance is considered at  $p$  values of  $<0.10$ . Analysis was conducted for the period 1827–1999 and also for the period 1974–1999 to eliminate the extreme influence of the anomalous “presettlement” vegetation upon the statistics.

### **Power Law**

The time series of land-cover data were examined for threshold effects. Milne (2000) predicted that if the base 10 logarithm of the number of patches in a particular land-cover class is plotted against the base 10 logarithm of the mean patch size for that class, then either: (1) a significant power law function may fit the relationship [Fig. 1(A)] or (2) breaks or disjunctions in the power law fit may be detected [Fig. 1(B)]. A significant power law indicates that both energy dissipation and scale invariance occur where either a set of constraints or a single constraint brings about the observed patch dynamics (Bak and Tang 1989; Ludwig et al. 2000; Milne 2000).

## **RESULTS**

The time series of filtered land-cover maps showed persistent landscape features as well as changes in land-cover composition through time (Fig. 3). While it was expected that the landscape had changed greatly between 1827 and 1974, the magnitude of change reflected may have been exaggerated due to the nature of the historical data. The rarity of deciduous, mixed forest and nonforest vegetation in 1827 may partly be attributed to the sampling scheme used to create the data and also to surveyor bias. Furthermore, we have equated “cleared or bare or developed areas” with the large Native American settlements superimposed on the historical map. In reality, there could have been many more of these areas, and they may be better characterized as “nonforest” than as “cleared” areas. The dominance of pine on the historical landscape, however, was supported by the fact that about 90% of the soils at Fort Benning can support longleaf pine (Dale et al. 2002). Class-level metrics that were deemed significant from the time-series analysis are presented below.

The time-series analysis showed that several metrics were significant for mixed forest from 1827 to 1999 (Table 3). As expected, less significant change occurred between 1974 and 1999 although changes in the similarity index were noted (Table 4). The mixed forest class displayed a power law and was found to be scale-invariant from 1827 to 1999 ( $r^2 = 0.98$ ) and from 1974 to 1999 ( $r^2 = 0.72$ ).

There were also significant trends for nonforest vegetation from 1827 to 1999 (Table 5). No significant trends were found from 1974 to 1999. Metrics for this class, however, displayed different characteristics in 1983 than for other years in the time series. After 1983, mean patch area decreased and the number of patches increased. Furthermore, the nonforest class had the largest mean patch size in 1983. It is possible that a single isolated land use could be responsible for this trend in the early 1980s. The nonforest class was found to be scale-invariant under power law analysis from 1827 to 1999 ( $r^2 = 0.98$ ) and from 1974 to 1999 ( $r^2 = 0.82$ ). The time-series analysis showed that several metrics were significant for pine forest between 1827 and 1999 (Table 6). The significance of these trends, however, was biased by the nearly continuous representation of pine forest in the 1827 map. No significant trends were present for pine forest from 1974 to 1999. Although the number of patches decreased after 1983, the mean patch size, patch shape simplicity, and the percentage of pine composition has increased. This difference could be the result of aggressive management for pine, including the establishment of pine



plantations. Pine forests displayed a power law and were found to be scale- invariant from 1827 to 1999 ( $r^2 = 0.997$ ) and from 1974 to 1999 ( $r^2 = 0.88$ ).

The time-series analysis for deciduous forest showed significant changes in several metrics from 1827 to 1999 (Table 7). No significant trends were present for deciduous forest from 1974 to 1999. It may be worth noting, however, that the number of deciduous patches decreased between 1991 and 1999, in

contrast to an earlier increase. Unlike the other land-cover classes considered in this research, the deciduous forest class did not display a power law function. This result implies that the deciduous forest has exceeded a threshold because the processes that led to its formation are in non-equilibrium (Fig. 4). From 1991 to 1999, patch number decreased and patch area remained the same. This pattern suggests that different processes are involved at varying spatial scales in the structuring of deciduous forests and could reflect evolving approaches to forest management.



The time-series analysis showed several metrics were significant for the bare/developed class between 1827 and 1999 (Table 8). Only the patch-area coefficient of variation (CV), however, was significant for the period between 1974 and 1999 (Table 9). The bare/developed class exhibited a power law. The data were found to be scale-invariant from 1827 to 1999 ( $r^2 = 0.996$ ) and a little less so from 1974 to 1999 ( $r^2 = 0.79$ ).

Class-level metrics were also reviewed outside the context of a time-series analysis to determine which classes exhibited unique or similar patch characteristics. For example, deciduous forest, bare/developed areas, and nonforest vegetation were all characterized by relatively low perimeter-area ratios. Deciduous forest is commonly associated with riparian areas, and bare/developed areas and nonforest vegetation are largely associated with managed land use and found in simple, confined shapes. Bare/developed patches were the class most widely dispersed on the landscape, followed by nonforest patches. These classes were largely associated with managed land uses and comprised a smaller percentage of the landscape and a smaller number of patches when compared to other classes. The mixed forest class was less clumped than other classes. This difference can be attributed largely to the transitional nature of this class. Furthermore, mixed forest was commonly found interspersed along other forest and nonforest patch boundaries instead of in continuous patches.

In addition to computing metrics for each land-cover class, we also examined landscape-level metrics. While landscape-level metrics do not give specific information about individual vegetation types, they can be useful indicators of overall landscape fragmentation and diversity. A time-series analysis of landscape-level metrics followed similar trends to those of the class-level metrics for the time period between 1827 and 1999 (Tables 10–13).



## **DISCUSSION**

Dramatic changes to the Fort Benning landscape occurred between 1827 and 1974. The landscape, once dominated by almost uninterrupted pine forest, transitioned into a mosaic of many different land uses and cover types, as has occurred elsewhere in the southeastern United States (Frost 1993). Several class- and landscape-level metrics support this trend towards a more fragmented landscape since 1827. These include significant decreases in mean patch area, clumpiness, and nearest neighbor distance for all classes. Consequently, indices such as edge, number of patches, patch-area coefficient of variation, perimeter/area ratio, and to a lesser degree shape, have all increased significantly since 1827.

While the methods we used to generate the land-cover surface for 1827 may have exaggerated the significance of these trends, changing land uses associated with settlement, agriculture, military training and development, and forest management have most certainly altered the landscape between 1827 and 1999.

The percentage of the landscape classified as bare/developed increased after 1827. In light of this increase, the values for patch area and clumpiness were contrary to what was expected. Patches of bare/developed areas decreased in area and clumpiness but increased in area variability and shape complexity between 1827 and 1999. These trends make sense when examined within the context of the methodology used to create the 1827 map. Large, known settlements were superimposed upon the 1827 land-cover map to account for bare/developed areas. This context is important for several reasons: (1) only the largest settlements, based upon archaeological evidence, were mapped (it is known that many more small settlement areas existed in the area that is now Fort Benning in 1827); (2) these areas were mapped as large, simplified, contiguous shapes; and (3) areas of settlement would likely have included both bare ground and cleared areas of successional, nonforest vegetation or even planted crops. It is expected, however, that the amount and nature of nonforested and bare ground changed dramatically following European settlement and the creation of Fort Benning.

When the conditions of 1827 were removed from the time-series analysis, few metrics showed significant trends between 1974 and 1999. Bare/developed patch area variability increased during this time period. Areas of bare ground were directly associated with human land use and have largely persisted on the landscape during recent decades. Tracked vehicles are especially detrimental to vegetation, which is one reason that training with tracked vehicles is restricted to certain areas. The concept is that it is better to the overall environment to sacrifice a few areas while maintaining high-quality ecological conditions in other areas. New areas of bare ground have not been widely introduced at Fort Benning during the recent past; however, patch areas expand and contract with periods of more intensive military training.

Mixed forest patches showed significant trends for a similarity index between 1974 and 1999, which could result from an anomalous peak in mixed forest pixels in 1983 due to misclassification errors. The effect of classification errors for this class, however, should be lessened by the use of the filtered data. This trend may therefore be indicative of the transitional nature of the mixed forest class, which includes successional environments as well as mixed forests.

While few metrics showed significant change between 1974 and 1999, some notable changes to land cover at Fort Benning occurred in recent decades. Several trends, such as an increase in nonforested and barren lands in riparian buffers, were slowed or reversed. While the percentage of nonforest land and the number of nonforest patches present have been increasing since 1974, the size of these patches has decreased. Though changes in these metrics over time were not statistically significant, it is worthwhile to recognize that concentrated management has likely constrained certain land uses to smaller geographic areas.

While clumpiness was an important indicator of change between 1827 and 1999, changes between 1974 and 1999 were not statistically significant. Clumpiness, however, was found to be a useful metric for differentiating between land-cover categories. For example, mixed forests were less aggregated than other classes, and bare/developed patches were more clumped.



Changes in forest dynamics have also been evident in recent decades. While other land-cover classes exhibited a power law function, deciduous forest displayed scale invariance during the time series. This result implies that thresholds had been exceeded between periods when the balance between constraints was altered, possibly by a change in or introduction to new land management practices. Between 1991 and 1999, the number of deciduous patches decreased while mean patch area remained constant. Improved monitoring techniques coupled with an aggressive management strategy for perpetuating pine forest at Fort Benning (Waring et al. 1990) have likely been the cause for an increase in pine populations and a decrease in hardwood invasion in the 1990s.

Specifically, the 1994 Fish and Wildlife Service Jeopardy Biological Opinion (JBO) had a tremendous impact upon forest management at Fort Benning (Rob Addington, personal communication, April 2004). Following the JBO, Fort Benning reformulated its management goals to emphasize reforestation of longleaf pine to the benefit of species such as the red-cockaded woodpecker. This management strategy includes harvesting timber from stands that are not longleaf pine, thinning and burning areas of longleaf pine, and planting areas in pine plantations (Gilliam and Platt 1999; Haywood et al. 1998; Provencher et al. 2001).

Examination of the time series of landscape-level metrics revealed a significant trend towards a more fragmented landscape. Patch density and edge metrics increased, while the size of the largest patch and measures of radius of gyration decreased, indicating a trend towards more, smaller patches on the landscape. Increases in shape metrics such as perimeter/area ratios and the fractal dimensions index indicate that shape complexity has increased significantly since 1827. Measures of isolation/proximity decreased since 1827. A similarity index increased while Euclidean nearest neighbor distance decreased. The landscape of 1827 was composed of a pine forest matrix that contained isolated, continuous patches of deciduous forest, mixed forest vegetation, and cleared areas. The landscape has evolved to a complex mosaic of different land uses and land cover. Land-cover classes are more like their neighbors, and the presence of transitional areas between classes also influences metrics of similarity. Metrics of contagion and interspersion reveal that the landscape has become less aggregated. A trend of decreasing contagion, the percentage of like adjacencies, and aggregation are indicative of disaggregation and dispersion. An increasing trend in patch division further supports landscape disaggregation. Connectivity has also decreased, as shown by a significant decrease in landscape cohesion. Diversity indices such as Simpson's diversity index and Shannon's evenness index also indicate the trend towards a more diverse landscape.

For the more recent time period, 1974–1999, few metrics displayed significant change through the time-series analysis. An increasing trend in the interspersion and juxtaposition index may be influenced by the transitional nature of successional environments. For example, a clearing in a pine forest would initially include only two land-cover classes, but after years of regrowth, several classes—bare ground, nonforest vegetation, mixed forest vegetation, pine, and even deciduous forest—could be present in that same area. Recent management efforts focused upon eliminating hardwood invasion into areas of longleaf pine may also be reflected here. The range of values for radius of gyration decreased between 1974 and 1999. This trend reveals that the discrepancy between the largest and smallest patches has decreased. A decrease in the size of the largest patches and/or an increase in size of the smallest patches could account for this trend. Since the largest patch index did not change significantly, a combination of the above effects is probable. The metrics on the whole do not reveal significant change at a landscape scale between 1974 and 1999.

The creation and selection of land-cover data were crucial aspects of this project. Even though the witness tree data for the southeastern states does not provide continuous coverage and is highly variable, it is useful for estimating land cover in the nineteenth century. By augmenting the witness tree data with archeological evidence, we gained greater confidence that the 1827 map reflects conditions at the time. For example, although much burning was conducted as part of land clearing by native populations in the central region of Georgia, the witness tree data confirms that pine forest still dominated the area. Similar witness tree data have been used to support the presence of extensive pine stands in frequently burned presettlement areas of the Missouri Ozarks (Batek et al. 1999) and northern Florida (Schwartz 1994).



While the anomalous conditions present in the 1827 map are real, the wide gap between the historical map and the satellite imagery of the 1990s caused some problems for interpretation of the trends. The characterization of additional time periods between 1827 and 1974 would be extremely useful in bridging the gap between the nineteenth-century landscape and the established military base of the 1970s. Project constraints did not allow for the inclusion of these data in this analysis. Work has already begun, however, to obtain and classify historical aerial photography from the 1940s and 1950s for selected areas of the base.

The selection of source imagery and land-cover data should be approached carefully. Map accuracy varies spatially and may be related to land-cover type, terrain, landscape complexity, and land use patterns (Steele et al. 1998). Furthermore, accuracy statements that accompany maps of large areas may be known to be erroneous, vague, or non-site-specific (Foody 2002). All data, therefore, should be reviewed carefully in the context of their intended use (Fosnight and Greenlee 2000). The minimum mapping unit employed can also have a dramatic impact upon the results of these types of analysis. Smoothing the classified imagery through a filter served to eliminate small land-cover patches, some of which may have been a result of classification errors. While metrics were computed for both filtered and unfiltered data, the time-series analysis was run only on the smoothed data to eliminate the impact of anomalous patches. Smoothing, however, may have exaggerated landscape and patch homogeneity. Reality exists somewhere between these two approaches, and the results, therefore, should be interpreted accordingly.

## **CONCLUSION**

Data collected for disparate purposes can be used to help develop an understanding of land-cover changes over time and are often necessary to further our knowledge of historic conditions on a given landscape. Examination of a time series of landscape metrics showed that the landscape as a whole, and several land-cover classes, exhibited significant change between 1827 and 1999. Overall, the metrics for 1827 were very different from the metrics representing recent decades. While the changes visible between 1827 and 1974 may be somewhat exaggerated due to data constraints, the nineteenth-century landscape at Fort Benning was composed largely of uninterrupted pine forest with some deciduous species found in riparian corridors and some open areas associated with Native American settlements. Land cover and land use in the 1970s and thereafter were considerably different. Decades of agricultural use and military training activities collectively worked to fragment the landscape. Deciduous species encroached on pine forests, and heavy training activities and construction resulted in areas of sparse land cover and bare ground.

In contrast, few significant changes were apparent between 1974 and 1999. Areas of bare ground persisted on the landscape throughout the 1980s and 1990s. This result not only emphasizes the lasting footprint these activities have on the landscape, but also highlights the efforts made by management to confine heavy training exercises to certain areas. The area of these patches has varied during periods of more intensive training and recovery. Furthermore, management efforts in the 1990s have focused upon practices that favor the establishment and maintenance of longleaf pine communities. These efforts appear to be decreasing hardwood invasion in favor of pine species in many areas on the installation.

Overall, examining a suite of landscape metrics over time was beneficial for summarizing, describing, and assessing recent and historical land-cover change at Fort Benning. More specifically, regression analysis was necessary for the determination of significant trends within the time series of landscape metrics. Testing for the presence of a power law function also proved helpful in identifying whether individual land-cover classes were in a state of disequilibrium. These methods, when used with a suite of landscape metrics, helped to identify and quantify land-cover change at Fort Benning between 1827 and 1999.



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## FIGURES

**Fig. 1.** Self-organized systems are scale-invariant—i.e., the same balance of system constraints exists at all observed scales (A). However, when the critical balance or poise between constraints is altered, the system can become disjoint and thus exceeds a threshold or thresholds (B).

**Fig. 2.** Fort Benning, Georgia, is located in southwestern Georgia near the town of Columbus, Georgia.

**Fig. 3.** The time series of standardized land-cover maps for (a) 1827, (b) 1974, (c) 1983, (d) 1991, and (e) 1999.

**Fig. 4.** The patch dynamics of the deciduous forest at Fort Benning, Georgia, from 1827 to 1999 were scale-invariant (A). However, the forest's dynamics from 1974 to 1999 indicate disjunction between both 1983 and 1991, and 1991 and 1999 (B). This implies that thresholds had been exceeded between these periods when the balance between constraints was altered.



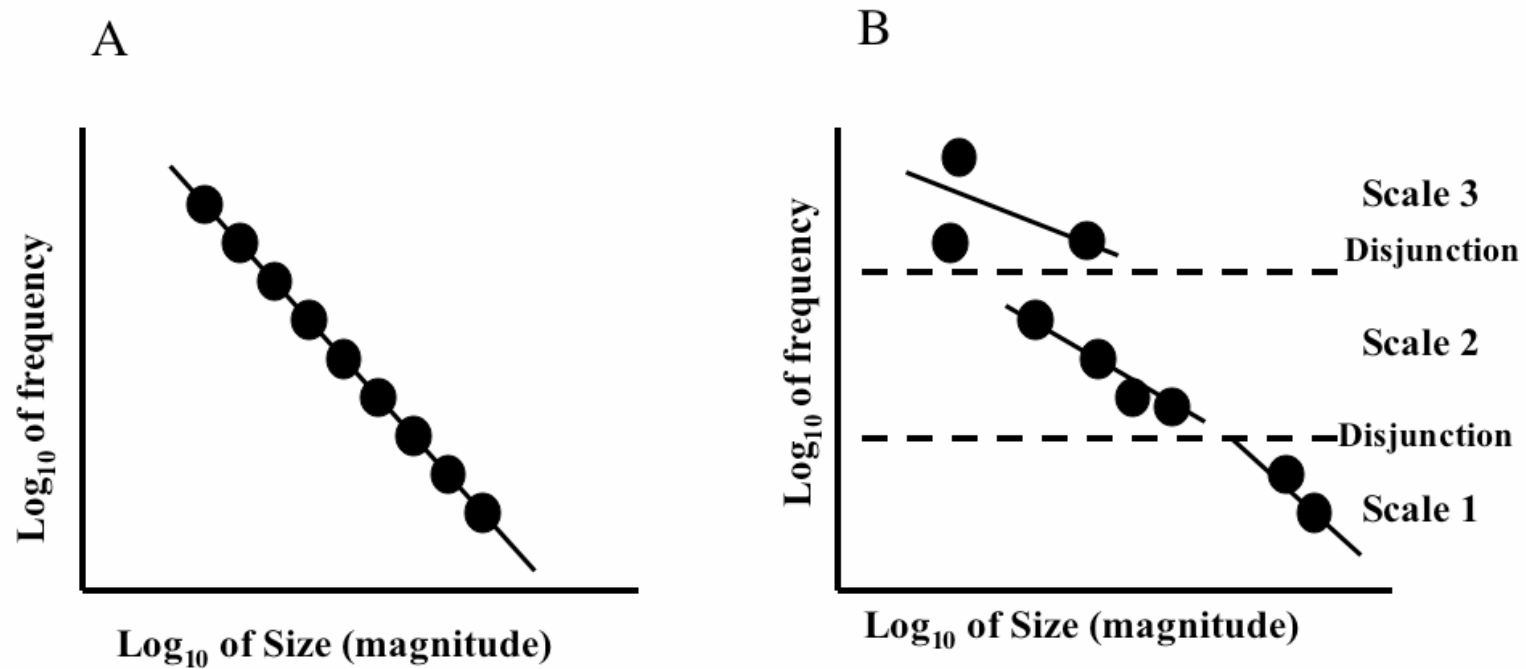
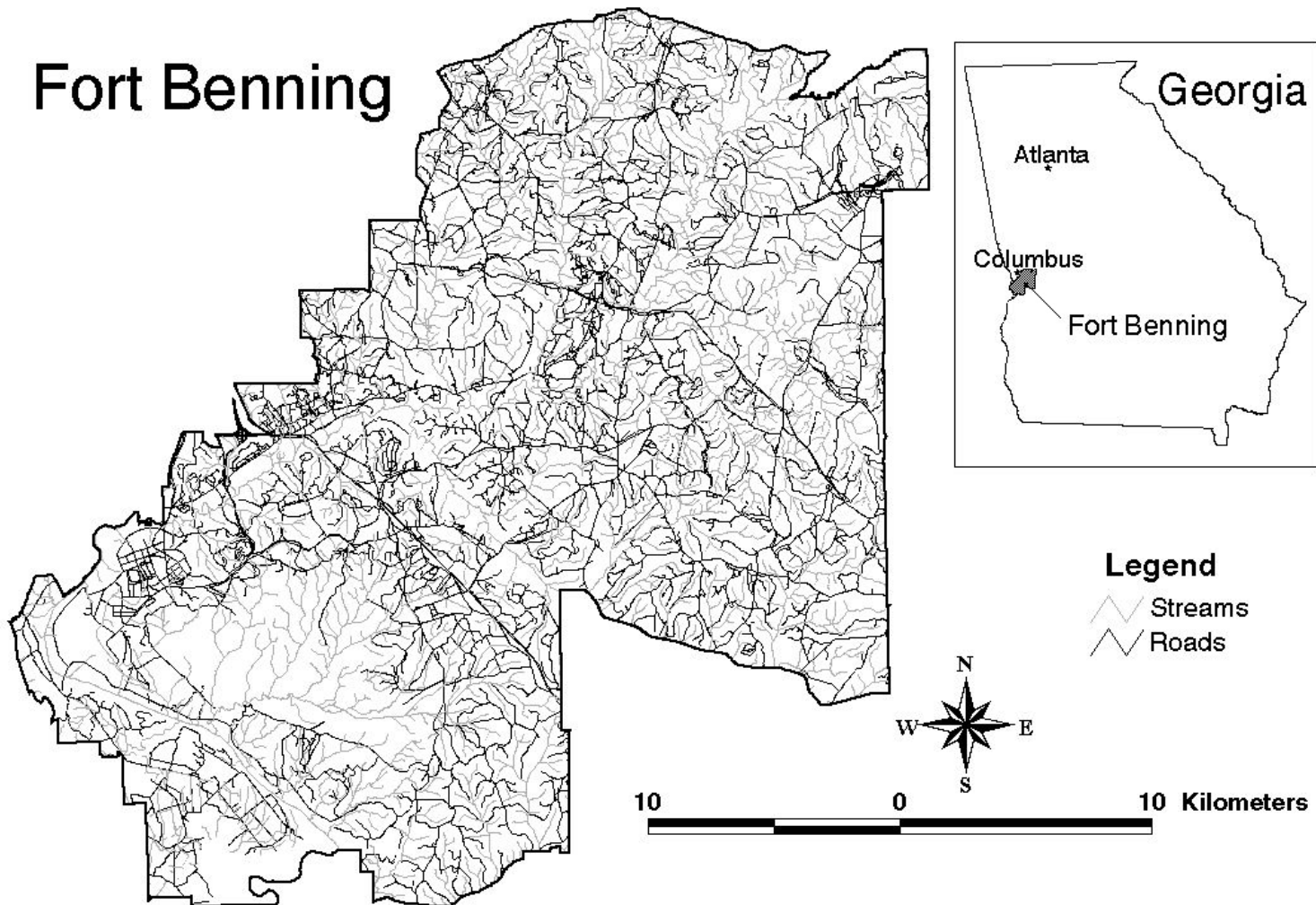


Fig.1

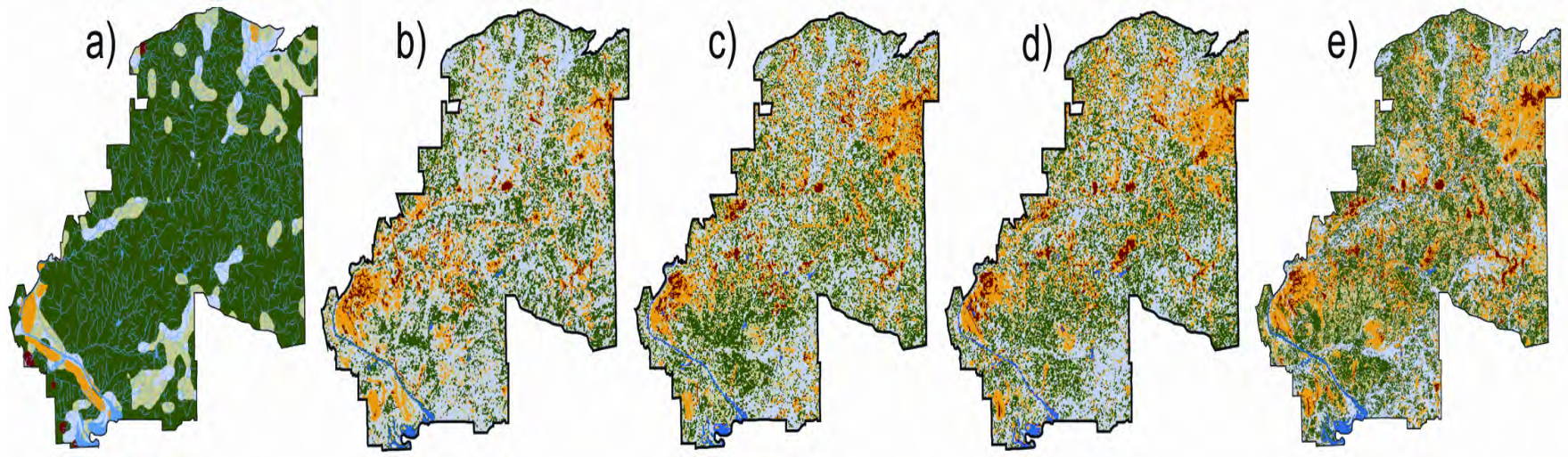


**Fig. 2**





**Fig. 3 Change in land cover over time: (a) 1827, (b) 1974, (c) 1983/1986, (d) 1991, and (e) 1999 where dark green = pine forest, light green = mixed forest, orange = cleared areas, red = bare ground, dark blue = water, and light blue =deciduous hardwood forest.**

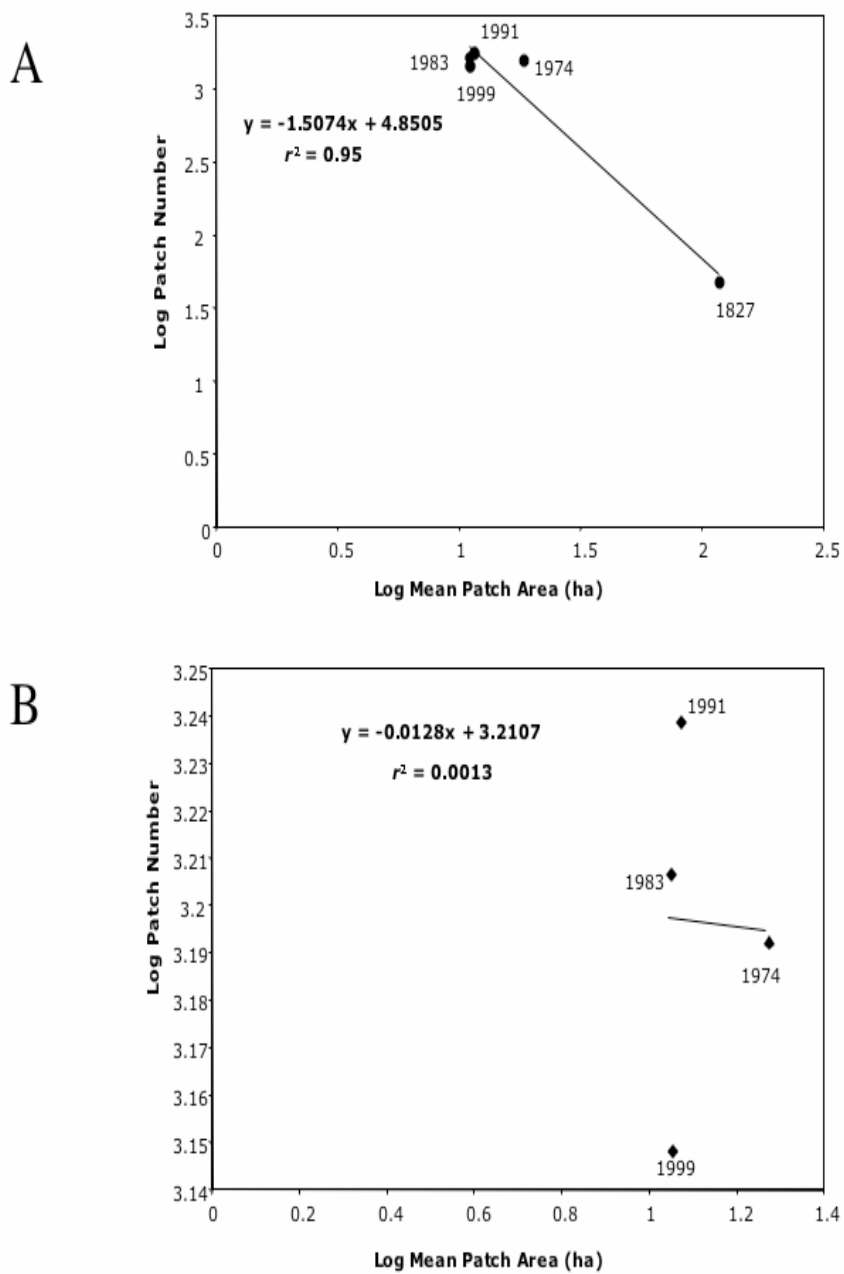








**Fig. 4**





## Tables

**Table 1. Data sources of the time series of land cover maps that were used in this study.**

| <b>Data Type</b>  | <b>Acquisition Date</b> | <b>Sources<sup>a</sup></b>           | <b>Resolution</b>   |
|---|-------------------------|--------------------------------------|---|
| Historical land cover interpolated from witness tree data | 1827                    | State of Georgia General Land Office | Clusters of points were roughly located every 0.5 miles. Gridded data were created at a 60-m resolution |
| Land cover  | 1974                    | Landsat MSS NALC                     | 60m   |
| Land cover  | 1983/1986               | Landsat MSS NALC                     | 60m   |
| Land cover  | 1991                    | Landsat MSS NALC                     | 60m   |
| Land cover  | 1999                    | Landsat 7 ETM+                       | 30m   |

<sup>a</sup> **MSS = Multispectral Scanner**; NALC = North American Land Characterization triplicate dataset; ETM+ = Enhanced Thematic Mapper Plus.



**Table 2.** Descriptions of selected landscape metrics.

| Metric type and importance   | What it measures   | Specific use in this study  |
|--|--|---|
| <b>Land cover</b><br>Used for calculating many other landscape-level indices   | Describes types of feature on the surface of the earth   | Continuous land cover surface generated using witness tree data for 1827. Land cover assigned to classifications of 4 image dates (1974, 1983/86, 1991, 1999).              |
| <b>Landscape composition</b><br>Useful for tracking large-scale changes in land cover through time   | Measures proportion of land cover types present within a landscape   | Landscape composition for 5 time periods, was summarized (barren or developed land, pine forest, deciduous forest, mixed forest, nonforest) using ATtILA/FRAGSTATS software |
| <b>Area/density/edge metrics</b><br>Quantify spatial heterogeneity and fragmentation on the landscape  | Measurement of total area association with each land-cover class, number of patches present for each land-cover class, patch density, sum of lengths (m) of all edge segments of a given land-cover class, edge density, size of largest patch associated with each land-cover class, distribution of distances between each cell in a patch and patch centroid. A <i>patch</i> is defined as an area of contiguous pixels of the same land-cover type   | Class area, number of patches, patch density, total edge, edge density, landscape shape index, largest patch index, radius of gyration distribution                         |
| <b>Shape metrics</b><br>Interaction of patch size and shape are important to several ecological processes; these metrics are important in determining the presence of edge effects | <i>Shape complexity</i> —measured by examining patch perimeter in relation to patch area; <i>shape index</i> —examines patch perimeter in relation to patch perimeter of a maximally compact shape of the same size  | Perimeter-area fractal dimension, perimeter-area ratio distribution, shape index distribution, fractal index distribution   |
| <b>Isolation/proximity metrics</b><br>Measures of patch isolation are important indicators of habitat fragmentation  | <i>Similarity index</i> —examines similarity between neighboring pixels (classes); <i>nearest neighbor distance</i> —measures shortest distance between two patches of same land-cover class   | Similarity index distribution, nearest neighbor distance distribution   |
| <b>Contagion/ interspersions metrics</b><br>Contagion and dispersion measure the dispersion and intermixing of patch types and reflect landscape texture                           | <i>Percentage of like adjacencies</i> —number of like adjacencies between pixels of a given class; <i>contagion</i> —considers proportional abundance of each patch type and proportion of adjacencies between patch types; <i>interspersions and juxtaposition index</i> —considers patch adjacencies instead of cell adjacencies; <i>clumpiness</i> —refers to proportional deviation of proportion of like adjacencies involving the corresponding class from that expected under a spatially random distribution | Percentage of like adjacencies, contagion, interspersions and juxtaposition index (IJI) and clumpiness  |
| <b>Connectivity metrics</b><br>Describe how well a landscape facilitates or impedes ecological flows   | Patch cohesion measures the connectivity of patch types by examining patch perimeter and area in a total landscape context   | Patch cohesion index  |



| Metric type and importance   | What it measures   | Specific use in this study |
|--|--|----------------------------|
| <b>Diversity indices</b><br>Include measures of richness (the number of patch types present) and evenness (the distribution of area among different types) | <i>Diversity indices</i> —record the proportion of the landscape occupied by each patch type; <i>evenness indices</i> —divide the proportion by number of patch types represented on the landscape |                            |

Source: McGarigal and Marks 1995.

**Table 3.** Significant trends in mixed forest metrics, 1827–1999.

| Metric            | Function | $r^2$ | p     | Trend       |
|-------------------|----------|-------|-------|-------------|
| Edge              | Linear   | 0.87  | 0.021 | Increase    |
| Number of patches | Linear   | 0.90  | 0.013 | Increase    |
| Mean patch area   | Linear   | 0.98  | 0.001 | Decrease    |
| Patch area CV*    | Linear   | 0.86  | 0.022 | Increase    |
| Perimeter/area    | Linear   | 0.98  | 0.001 | Increase    |
| Shape             | Linear   | 0.745 | 0.059 | Increase    |
| Similarity        | Linear   | 0.96  | 0.003 | Increase    |
| Clump             | Cubic    | 0.99  | 0.009 | Inc/dec/inc |
| Nearest neighbor  | Linear   | 0.94  | 0.007 | Decrease    |

\*CV = coefficient of variation.



**Table 4.** Significant trends in mixed forest metrics, 1974–1999

| <b>Metric</b> | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|---------------|-----------------|-------------------------|----------|--------------|
| Similarity    | Quadratic       | 0.84                    | 0.081    |              |

**Table 5.** Significant trends in nonforest vegetation metrics, 1827–1999.

| <b>Metric</b>     | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|-------------------|-----------------|-------------------------|----------|--------------|
| Edge              | Linear          | 0.85                    | 0.025    |              |
| Number of patches | Linear          | 0.92                    | 0.01     | Increase     |
| Mean patch area   | Linear          | 0.98                    | 0.001    | Decrease     |
| Patch area CV*    | Linear          | 0.92                    | 0.01     | Increase     |
| Perimeter/area    | Linear          | 0.99                    | <0.001   | Increase     |
| Shape             | Linear          | 0.81                    | 0.036    | Increase     |
| Similarity        | Linear          | 0.99                    | 0.001    | Increase     |
| Clump             | Linear          | 0.98                    | 0.001    | Decrease     |
| Nearest neighbor  | Linear          | 0.98                    | 0.001    | Decrease     |

\*CV = coefficient of variation.



**Table 6.** Significant trends in pine forest metrics, 1827–1999.

| <b>Metric</b>     | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|-------------------|-----------------|-------------------------|----------|--------------|
| Edge              | Linear          | 0.95                    | 0.024    | Increase     |
| Number of patches | Linear          | 0.96                    | 0.02     | Increase     |
| Mean patch area   | Linear          | 0.99                    | 0.004    | Decrease     |
| Patch area CV*    | Linear          | 0.99                    | 0.006    | Increase     |
| Perimeter/area    | Linear          | 0.99                    | 0.002    | Increase     |
| Similarity        | Linear          | 0.96                    | 0.021    | Increase     |
| Clump             | Linear          | 0.999                   | 0.001    | Decrease     |
| Nearest neighbor  | Linear          | 0.998                   | 0.001    | Decrease     |

\*CV = coefficient of variation.

**Table 7.** Significant trends in deciduous forest metrics, 1827–1999.

| <b>Metric</b>       | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|---------------------|-----------------|-------------------------|----------|--------------|
| Edge                | Linear          | 0.99                    | 0.005    | Increase     |
| Number of patches   | Linear          | 0.98                    | 0.01     | Increase     |
| Mean patch area     | Linear          | 0.99                    | 0.003    | Decrease     |
| Patch area CV*      | Linear          | 0.86                    | 0.072    | Increase     |
| Perimeter/area      | Linear          | 0.99                    | <0.001   |              |
| Similarity          | Linear          | 0.95                    | 0.025    | Increase     |
| Clump               | Linear          | 0.99                    | 0.002    | Decrease     |
| Nearest neighbor CV | Linear          | 0.99                    | 0.005    | Decrease     |

\*CV = coefficient of variation.



**Table 8.** Significant trends in bare/developed metrics, 1827–1999.

| <b>Metric</b>       | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|---------------------|-----------------|-------------------------|----------|--------------|
| Edge                | Linear          | 0.90                    | 0.013    | Increase     |
| Number of patches   | Linear          | 0.93                    | 0.007    | Increase     |
| Mean patch area     | Linear          | 0.98                    | 0.001    | Decrease     |
| Patch area CV*      | Cubic           | 0.96                    | 0.035    | Increase     |
| Perimeter/area      | Linear          | 0.99                    | <0.001   | Increase     |
| Similarity          | Linear          | 0.95                    | 0.004    | Increase     |
| Nearest neighbor CV | Linear          | 0.99                    | <0.001   | Decrease     |
| Clump               | Linear          | 0.99                    | 0.001    | Decrease     |

\*CV = coefficient of variation.

**Table 9.** Significant trends in bare/developed metrics, 1974–1999.

| <b>Metric</b>  | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|----------------|-----------------|-------------------------|----------|--------------|
| Patch area CV* | Linear          | 0.97                    | 0.013    | Increase     |

\*CV = coefficient of variation.

**Table 10.** Trend of landscape-level metrics of Fort Benning from 1827 to 1999.

| <b>Metric</b>       | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|---------------------|-----------------|-------------------------|----------|--------------|
| Number of patches   | Linear          | 0.93                    | 0.008    | Increase     |
| Patch density       | Linear          | 0.93                    | 0.008    | Increase     |
| Largest patch       | Linear          | 0.99                    | 0.000    | Decrease     |
| Edge                | Linear          | 0.945                   | 0.005    | Increase     |
| Nearest neighbor    | Linear          | 0.94                    | 0.007    | Decrease     |
| Area                | Linear          | 0.98                    | 0.001    | Decrease     |
| Radius—gyration     | Linear          | 0.98                    | 0.001    | Decrease     |
| Shape               | Linear          | 0.86                    | 0.022    | Decrease     |
| Fractal (para)*     | Linear          | 0.99                    | 0.000    | Increase     |
| Contagion           | Linear          | 0.98                    | 0.001    | Decrease     |
| Cohesion            | Linear          | 0.80                    | 0.04     | Decrease     |
| Division            | Linear          | 0.98                    | 0.001    | Increase     |
| Shannon's diversity | Linear          | 0.99                    | 0.000    | Increase     |
| Simpson's diversity | Linear          | 0.99                    | 0.001    | Increase     |
| Shannon's evenness  | Linear          | 0.99                    | 0.001    | Increase     |
| Simpson's evenness  | Linear          | 0.99                    | 0.001    | Increase     |

\* Fractal (para) is the perimeter-area fractal dimension.



**Table 11.** Trend of fractal statistics at the landscape-level from 1827 to 1999.

| <b>Metric*</b> | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|----------------|-----------------|-------------------------|----------|--------------|
| Fractal_am     | Linear          | 0.90                    | 0.014    | Increase     |
| Fractal_md     | Linear          | 0.68                    | 0.085    | Decrease     |
| Fractal_ra     | Linear          | 0.89                    | 0.016    | Increase     |
| Fractal_sd     | Linear          | 0.90                    | 0.013    | Increase     |
| Fractal_cv     | Linear          | 0.91                    | 0.011    | Increase     |

\*am = area-weighted mean; md = median; ra = the range; and sd = standard deviation.

**Table 12.** Trend of similarity statistics at the landscape-level from 1827 to 1999.

| <b>Metric*</b> | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|----------------|-----------------|-------------------------|----------|--------------|
| Simil_am       | Linear          | 0.64                    | 0.100    | Increase     |
| Simil_md       | Linear          | 0.71                    | 0.074    | Increase     |
| Simil_ra       | Linear          | 0.75                    | 0.059    | Increase     |
| Simil_cv       | Linear          | 0.99                    | 0.000    | Decrease     |

\*am = area-weighted mean; md = median; ra = the range; and sd = standard deviation.

**Table 13.** Trend of landscape-level metrics of Fort Benning from 1974 to 1999. The suffix ra = the range of the radius of gyration distribution. IJI is the Interspersion/Juxtaposition Index.

| <b>Metric*</b> | <b>Function</b> | <b><math>r^2</math></b> | <b>p</b> | <b>Trend</b> |
|----------------|-----------------|-------------------------|----------|--------------|
| Gyrate_ra      | Linear          | 0.85                    | 0.076    | Decrease     |
| IJI            | Linear          | 0.87                    | 0.66     | Increase     |

\*ra = the range; IJI = Interspersion/Juxtaposition Index.



## Section 7. Watershed Indicators

Streams are a good indicator of disturbances at the watershed scale because drainage water integrates landscape-level processes occurring within watersheds. Therefore, to evaluate watershed scale indicators of disturbance we studied twelve 2<sup>nd</sup>- and 3<sup>rd</sup>-order streams in the eastern part of the Fort Benning Military Installation (FBMI) that drained watersheds with a wide range of disturbance levels. Study streams were low-gradient (range = 0.8 – 5.1%, mean = 1.9%) with an intact riparian canopy (summer canopy cover range = 89 – 96%, mean = 94%), typical of other sandy Southeastern Plains streams. Dominant riparian vegetation included blackgum (*Nyssa sylvatica*), sweetgum (*Liquidambar styraciflua*), and sweetbay (*Magnolia virginiana*). We quantified watershed disturbance level as the sum of (1) proportion of the watershed occurring as bare ground on slopes >3% and (2) proportion occurring as unpaved road cover. The details of this quantitative approach as a measure of watershed disturbance are provided in the manuscript in Appendix A (Maloney et al. in press). Study streams drained watersheds ranging in disturbance level from about ~2 to 14%. We then compared a variety of stream physical, chemical, and biological characteristics across this disturbance gradient to evaluate their usefulness as ecological indicators of disturbance.

Stream physical, hydrological, and organic variables were examined from 2000–2002 in all 12 streams (Appendix A, Maloney et al. in press). Physical variables measured were streambed stability and particle size, whereas organic variables included benthic particulate organic matter (BPOM), dissolved organic carbon (DOC), and coarse woody debris (CWD). For stream hydrology we estimated stream responsiveness to a storm event (hereafter “flashiness”). Results suggest that all physical, hydrological, and organic variables decreased with increasing watershed disturbance. However, BPOM and CWD showed the strongest relationship and therefore were the most consistent indicators of disturbance.

Stream chemical variables evaluated included inorganic and organic suspended sediment concentrations, pH, specific conductance, nutrient ( $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , and soluble reactive phosphorus [SRP]) and DOC concentrations, and major base cations and acid anions. Stream chemistry was evaluated under both baseflow and stormflow conditions. The results of this evaluation are provided in the manuscript in Appendix B (Houser et al., submitted). Results indicate that total and inorganic suspended sediments, SRP, and DOC concentrations at baseflow and total and inorganic suspended sediments, SRP, and  $\text{NO}_3^-$  concentrations during storms were good indicators of disturbance.

Stream diurnal dissolved oxygen profiles were evaluated as an indicator of watershed-scale disturbance because these can provide a measure of metabolism within streams. The results of this evaluation are provided in the manuscript in Appendix C (Mulholland et al., submitted). Results indicate that the daily amplitude of the diurnal dissolved oxygen deficit profile was highly correlated with daily rates of in-stream gross primary production and was a useful indicator of disturbance. Results also show that the daily maximum dissolved oxygen deficit was highly correlated with daily rates of in-stream respiration and was a useful indicator of disturbance.



Benthic macroinvertebrate communities were evaluated as an indicator of watershed-scale disturbance. We sampled macroinvertebrates in 7 streams seasonally from 2000-2003 using a combination of Hester-Dendy multiplate units and D-frame sweep nets taken from a range of stream microhabitats. Results of this evaluation are provided in the manuscript in Appendix D (Maloney et al., submitted). Our results suggest that irrespective of season, several richness measures (e.g., number of Ephemeroptera, Plecoptera, and Trichoptera [EPT] taxa and richness of Chironomidae) negatively corresponded with watershed disturbance, however, except for chironomid richness, all measures showed high among-season and inter-annual variation. Compositional and macroinvertebrate functional feeding group measures also showed high seasonal and annual variation, with only the % of macroinvertebrates clinging to benthic habitats (= % clingers) corresponding with disturbance. Both tolerance metrics tested, the Florida Index and North Carolina Biotic Index, showed little seasonal and annual variation, however only the Florida Index was correlated with disturbance. A regional multimetric index, the Georgia Stream Condition Index (GASCI), was consistently related to watershed disturbance and also showed the least amount of seasonal and inter-annual variability. Our results further indicated a disturbance threshold at 8 to 10% of the watershed as bare ground and unpaved road cover, a threshold similar to that reported for other land uses.

Fish communities were evaluated as an indicator of watershed-scale disturbance. We sampled fish in 7 streams seasonally in 2003 using a backpack electroshocker and block seines (Appendix E, Maloney et al., submitted). As a result of low diversity traditional community-level measures of integrity (e.g., richness, diversity) were inappropriate, therefore we focused on 2 numerically dominant fish populations, the broadstripe shiner (Pteronotropis euryzonus), a vulnerable population, and the Dixie chub (Semotilus thoreauianus), a currently stable population. Proportions of these populations were strongly but oppositely associated with disturbance, with % of the assemblage as P. euryzonus and S. thoreauianus being negatively and positively related to disturbance, respectively. This complementarity likely resulted from contrasting life history traits, feeding behaviors, and/or habitat preferences between the 2 populations. Additionally, average size of both populations was lower in high-disturbance than low-disturbance streams, suggesting both populations were impacted by disturbance. Our results also suggested a disturbance threshold, where streams in watersheds with disturbance levels ~5 to 8% of the watershed disturbed had P. euryzonus proportions below those in low-disturbance streams.

Our final investigation dealing with stream indicators of disturbance involved an assessment of the relative importance of historical and contemporary disturbances. FBMI streams are influenced by both historic and contemporary watershed conditions, so we examined the relationships between contemporary stream physicochemical and biotic conditions and historic (1944) versus contemporary (1999) land use variables (Appendix F, Maloney et al., in preparation). Historic land use reflects disturbance from agriculture prior to military purchase, whereas contemporary land use reflects watershed disturbance resulting from a combination of military training and forest management practices. Contemporary streamwater chemistry (pH, SRP), organic matter variables (BPOM, CWD, DOC), and hydrologic flashiness were more related to contemporary than historic land use patterns, whereas streambed instability was strongly related to historic land use.



Stream diatom measures (% of diatoms as acidobiontic taxa and % of diatoms as motile taxa) were equally related to both contemporary and historic disturbance, suggesting diatoms were influenced by both historic and contemporary land use. Macroinvertebrate measures in which chironomids were highly influential (i.e., total taxa, diversity, number chironomid taxa) were best explained by contemporary land use. Macroinvertebrate measures incorporating larger and longer-lived taxa (EPT, % clingers, density) and fish assemblage measures (total taxa, diversity, total number collected) all were related to historic land use, collectively indicating that larger and longer-lived species may have been more susceptible to historic land use (i.e., potentially showing slower recovery rates) than smaller, short-lived organisms. Whole-stream respiration was related to contemporary land use, whereas gross primary production was related to historic land use and watershed size. Results suggest that contemporary stream patterns may be products of historic as well as, or in lieu of, contemporary land use conditions. Therefore, land use legacies at FBMI should be taken into consideration when examining the efficacy of ecological measures to indicate contemporary land use conditions and disturbance.



RH: Land use & stream physical and organic variables

**Influence of catchment-scale military land use on stream physical and organic matter variables in small Southeastern Plains catchments (USA)**

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Abstract. We conducted a 3-y study designed to examine the relationship between disturbance from military land use and stream physical and organic matter variables within 12 small (<5.5 km<sup>2</sup>) Southeastern Plains catchments at the Fort Benning Military Installation, Georgia, USA. Primary land-use categories were based on percentages of bare ground and road cover and nonforested land (grasslands, sparse vegetation, shrublands, fields) in catchments, and natural catchments features, including soils (% sandy soils) and catchment size (area). We quantified stream flashiness (determined by slope of recession limbs of storm hydrographs), streambed instability (measured by relative changes in bed height over time), organic matter storage (coarse wood debris [CWD] relative abundance, benthic particulate organic matter [BPOM]) and streamwater dissolved organic carbon concentration (DOC). Stream flashiness was positively correlated with average storm magnitude and % of the catchment with sandy soil, whereas streambed instability was related to % of the catchment containing nonforested (disturbed) land. Proportion of instream CWD and sediment BPOM, and streamwater DOC were negatively related to the % of bare ground and road cover in catchments. Collectively, our results suggest that the amount of catchment disturbance causing denuded vegetation and exposed, mobile soil is 1) a key terrestrial influence on stream geomorphology and hydrology, and 2) a greater determinant of instream organic matter conditions than is natural geomorphic or topographic variation (catchment size, soil type) in these systems.

Key words: land use, military training, landscape, stream, streambed instability, BPOM, coarse woody debris, streambed particle size.



Stream ecosystems are tightly coupled with their catchments, historically being a  
25 product of natural geological and climatological attributes. However, in many  
geographic regions dramatic human alteration of catchments from land-use practices  
has become a major source of landscape change (Hooke 1994, 1999). The effect of  
anthropogenic land use on stream systems is especially evident in the Southeastern  
Plains ecoregion of the United States (*sensu* Omernik 1987), where extensive  
30 agriculture, timber harvesting, and population growth have accelerated forestland  
conversion over the last 200 y (Hilliard 1984, USDOC 1990, Frost 1993). In addition,  
highly erodable sandy soils of this region (Griffith et al. 2001) coupled with extensive  
land use change may cause high upslope erosion and downslope sedimentation, and  
hence sediment delivery to receiving streams.

35 Unlike that of the Southeastern Plains, effects of land use on streams in upland  
regions have been extensively studied (Harmon et al. 1986, Herlihy et al. 1998, Paul  
and Meyer 2001, Meador and Goldstein 2003). For example, in streams within nearby  
upland Piedmont and Blue Ridge ecoregions, discharge and flashiness (*i.e.*, magnitude  
of hydrologic response to storms) often increase in response to catchment urbanization  
40 (Paul and Meyer 2001, Rose and Peters 2001) and forest harvest (Swank et al. 2001),  
as does increased export of dissolved organic carbon (DOC, Meyer and Tate 1983) and  
inorganic sediment loading from road construction (Swank et al. 2001, King and Gonsier  
1980, Reid and Dunne 1984). Unfortunately, extrapolation of changes in stream  
conditions associated with land use from stony high-gradient systems to those of sandy,  
45 low-gradient Southeastern Plains streams, especially concerning sediment movement  
and bed stability, may not be applicable (Feminella 2000). If true, then the inherent



difficulty in separating natural geomorphic influences from anthropogenic impacts on physicochemical and biotic variables within Southeastern Plains streams may be especially problematic (but see Morgan and Good 1988, Lenat and Crawford 1994, Dow  
50 and Zampella 2000).

Military installations occur throughout the United States and generally contain large tracts of land devoid of contemporary urban or agricultural land use. Streams draining military lands often are exposed to catchment disturbance from recurring training maneuvers ranging from light dismounted infantry and mechanized forces to  
55 munitions detonation and use of heavy (tracked) vehicles (Dale et al. 2002). The spatial extent of training and associated disturbance ranges from localized to broad scale, where loss of vegetation, soil compaction, and sediment runoff can occur over several contiguous hectares (Goran et al. 1983, Shaw and Diersing 1990, Milchunas et al. 2000). Such large-scale disturbances are similar to forest land clearing for suburban,  
60 urban, and agricultural development, in terms of increased soil erosion and sedimentation in streams (Howarth et al. 1991, Quist et al. 2003). However, whereas physical disturbance to the soil surface from suburban, urban, and agricultural development may be short term (months to years), lasting only until denuded soils are stabilized by revegetation and/or physical remediation, disturbance from military training  
65 often is continuous (decades) from repeated training of military personnel. Thus, compared with other land uses, catchments within military installations may be subjected to prolonged, repeated surficial soil disturbance, which may have pervasive impacts on streams and their biota.



To date, the majority of stream studies conducted on military lands have been  
70 focused on developing bioassessment protocols (e.g., Tertuliani 1999, Gregory et al.  
2001) rather than explicitly addressing nutrient and sediment runoff into receiving  
streams associated with training. To our knowledge, only Quist et al. (2003) addressed  
putative impacts of military training on stream abiotic factors, reporting increased  
sediment in stream pools and riffles and associated increases in silt-tolerant fauna,  
75 within high-use catchments. However, that study was done in mesic tallgrass prairie  
systems containing naturally steep-gradient streams with coarse substrate; virtually  
nothing is known about the degree to which landscape alteration from military training  
affects stream physical and organic matter variables in relatively low-gradient, sandy  
channels such as in the Southeastern Plains.

80 We investigated the relationships between military land use and physical and  
organic matter variables within small Southeastern Plains streams at the Fort Benning  
Military Installation, Georgia. Specifically, we tested whether stream hydrology  
(discharge, flashiness), geomorphology (streambed instability), and organic matter state  
(dissolved organic C, coarse woody debris, benthic particulate organic matter  
85 abundance) were better explained by variation in military land use at the catchment  
scale versus that of natural catchment features, including drainage area and  
predominant soil types.



## Methods

### Study site

We studied several streams and their catchments at the Fort Benning Military Installation (FBMI), within the Middle Chattahoochee River Drainage, in west-central Georgia (Figure 1). Catchments were within the Southeastern Plains ecoregion and the Sand Hills and Southern Hilly Gulf Coastal Plain subcoregions (Griffith et al. 2001). Fort Benning comprises 73.7 km<sup>2</sup> and is possibly an important source of sedimentation in the Chattahoochee River Basin (NAPA 2001). Prior to military acquisition, the primary land use at FBMI was row crop agriculture and pasture (Kane and Keeton 1998, USAIC 2001). The US military purchased ~50% of the present-day area in 1918 and the remainder in 1941 and 1942. Since the 1940s FBMI has been used for infantry and mechanized training with associated heavy equipment vehicles including tanks, armored personnel carriers, and a variety of light- and heavy-wheeled vehicles (USAIC 2001).

At FBMI, military training dramatically alters the landscape by disrupting vegetative cover and the underlying surface soil layer (Dale et al. 2002). Training maneuvers are localized and contained within compartments and differ in size and magnitude depending on compartment, thus it was possible to investigate the effects of catchment-scale disturbance across a range of disturbance intensities. Additionally, forestry practices such as controlled burning and timber harvesting frequently are used at FBMI (USAIC 2001), in large part relating to reestablishment of longleaf pine (Pinus palustris) forests and endemic red-cockaded woodpecker (Picoides borealis) populations (Dale et al. 2002, see also Noss 1989).



We studied twelve 2<sup>nd</sup>- and 3<sup>rd</sup>-order stream catchments on the eastern part of FBMI (Figure 1). Vegetation was primarily oak-hickory-pine and southern mixed forest, with underlying sandy or sandy clay loams soils (Omernik 1987, Griffith et al. 2001).

Dominant soil series found within catchments were Troup, Lakeland, Nankin and

115 Cowarts soils. Troup soils are deep, excessively drained, moderately permeable loamy, kaolinitic, thermic Grossarenic Kandiudults. Lakeland soils are very deep, excessively drained, rapidly to very rapidly permeable, thermic coated Typic Quartzipsamments.

Nankin soils are very deep, well drained, moderately slowly permeable, fine, kaolinitic, thermic Typic Kanhapludults. Dominant hydric soils included Bibb and Chastain soils.

120 Bibb soils are very deep, poorly drained, moderately permeable coarse-loamy, siliceous, active, acid, thermic Typic Fluvaquents. Chastain soils are very deep, poorly drained, fine, mixed, semiactive, acid, thermic Fluvaquentic Endoaquepts (Soil Survey Staff ). Geology is mainly Cusseta Sand (Lawton 1976). Study streams were low-

gradient (range = 0.8 – 5.1%, mean = 1.9%) with an intact riparian canopy (summer

125 canopy cover range = 89 – 96%, mean = 94%, Table 1), typical of other sandy Southeastern Plains streams (Felley 1992, Griffith et al. 2001). Dominate riparian vegetation included blackgum (Nyssa sylvatica), sweetgum (Liquidambar styraciflua), and sweetbay (Magnolia virginiana, Cavalcanti 2004).

### 130 Land use classification

We quantified land use within study catchments using geographic information system (GIS) datasets (i.e., streams: 1:24,000, 1993 coverage; soils: 1:20,000, 1998; roads: 10-m resolution, 1995), digital orthophotographs (1:5,000, July 1999), digital



elevation models (DEMs, 1:24,000, 10-m grid size, 1993) and Landsat imagery (28.5 m, July and December 1999). We processed data sets with catchment boundaries using ArcView© software (Environmental Systems Research Institute, Inc., Redlands, California). Landsat imagery, digital orthophotography, and DEMs were used to quantify the proportion of each catchment occurring in a particular land-use class on slopes >3%, using the ArcView extension Analytical Tools Interface for Landscape Assessments (ATtILA, Ebert and Wade 2000). We used 3% slopes as our threshold value because examination of the relationship between the calculated universal soil loss equation and catchment slope indicated that slopes at or above this level showed the highest potential for increased annual soil loss in our study area (see GASWCC 2000).

Land-use and land cover categories used in our analyses (Figure 2) were the proportion of bare ground and unpaved road cover (**%BGRD**) and the proportion of nonforested land in a catchment (**%NF**), whereas natural geomorphic categories included catchment size (**Area**), and the proportion of the catchment containing sandy, erodible soils (**%Sand**). The proportion of catchment on soils with >3% slopes and containing no vegetative cover was included in %BGRD (Figure 2A), which also included unpaved roads. We quantified road cover by multiplying road length by average width; the latter was estimated in the field for the 2 different classes of unpaved roads found in our catchments: class-6 roads (6-m wide, Figure 2B), and class-5 roads (20-m wide, Figure 2C). The proportion of catchment on soils with >3% slopes that were vegetated but without dense forests, including grasslands, sparse vegetation, shrublands, and fields was incorporated into %NF (Figure 2D). Unfortunately, we were unable to separate forest harvesting practices from other types of non-forested land as



a result of the low intensity of the selective cutting coupled with the resolution of land use data (30m). The proportion of catchment on Ailey loamy coarse sand and Lakeland sand soils was included in %Sand. Other soil types (sandy clay loams, loamy sands) were highly correlated with %Sand, so we excluded them from analysis. We defined Area as the catchment size (km<sup>2</sup>) drained by the study stream upslope of our sampling location, determined using DEMs and ArcView. The proportion of forested land in catchments was highly negatively correlated with %NF ( $r = -0.91$ ,  $p < 0.0001$ ) and %BRGD ( $r = -0.69$ ,  $p = 0.006$ ), so we excluded this variable from analysis.

#### Physical and organic matter variables

Streambed instability.—We estimated streambed instability by quantifying sediment movement using a modified transect method (Ray and Megahan 1979). We established cross-stream transects ( $n = 5$  per stream) by staking pairs of rebar on opposite banks of the channel perpendicular to flow. We leveled each transect (using a line level) and marked leveled heights on rebar pieces with cable ties. We quantified streambed height along fixed points of transects (20-cm intervals) by measuring vertical distance between the stream bottom and a fiberglass tape stretched across the channel; measures were made initially in January 2003 and then in July 2003 (~7 mo interval). Several storm events occurred during this sampling interval (KOM, unpublished data), so we considered this period sufficient to characterize relative changes in bed height among streams. We calculated streambed instability as the average absolute difference in height for each transect over the sampling period.



180        Stream flashiness.— Storm hydrograph recessions integrate numerous sources  
of inflow (e.g., overland flow, interflow), and have been used by others to indicate  
stream flashiness (Rose and Peters 2001). Therefore, we quantified the rate of  
descent of the falling limb of several storm hydrographs in each stream as a relative  
measure of hydrologic flashiness. We estimated discharge from measurements of  
185    channel width, and water velocity and depth measured by an ISCO ultrasonic flow  
module (model 750) and series portable sampler (model 6700); depth and velocity were  
recorded every 30 min to 1 h. We computed recession constants for the initial portion  
of the hydrograph recession curves for each storm hydrograph as the slope of the  
natural logarithm of discharge over time during the first 4 h following peak discharge  
190    (see Rose and Peters 2001). If the recession limb showed an obvious break in slope in  
<4 h, then we used data only prior to the break point to calculate recession constants.  
As a measure of storm magnitude we calculated the ratio of maximum discharge to prior  
baseflow discharge for each storm event [ $\max(Q/Q_{\text{base}})$ ]. We only included storms with  
 $\max(Q/Q_{\text{base}}) > 4$  because smaller storms did not have well-defined storm hydrographs  
195    (KOM, unpublished data). For 3 study catchments (LPK, SB1, SB5), we collected data  
for <3 storms, so we excluded these sites from analyses.

BPOM, particle size, and coarse woody debris.—We used sediment cores (PVC  
pipe, area =  $2.01 \text{ cm}^2$ , 10-cm depth) to quantify proportion of benthic particulate organic  
200    matter (BPOM) and streambed particle size. We considered BPOM all organic matter  
material  $\leq 1.6 \text{ cm}$  diameter, and quantified BPOM at 3 sites per stream every 2 mo  
(August 2001 to May 2003) and streambed particle size every 4 mo (September 2001 to



May 2002). For BPOM analysis, we took 3 cores from the stream thalweg, oven-dried each sample at 80°C for 24 to 48 h, and then weighed it. Samples were then ashed in a muffle furnace at 550°C for 3 h, cooled in a desiccator, and reweighed; % BPOM was determined as the difference between dry and ashed masses divided by total dry mass. For particle size analysis, we collected 2 cores per site, 1 in the thalweg and 1 near the stream margin. We combined cores within each site ( $n = 3$ ), removed organic matter and dispersed particles following the pipette method from a 10-g subsample (Gee and Bauder 1986). Particle sizes were then separated by dry sieving (2.0, 1.0, 0.5, 0.250, 0.125, 0.063 and <0.053 cm fractions), and mean weighted particle size for each stream was estimated by multiplying the mass of each fraction by the midpoint between sieve fractions and then dividing by the total sample weight. Particles >2 mm were removed prior to the dispersing process (see Gee and Bauder 1986). However, we estimated the % of the entire sample that was >2 mm prior to dispersion and used this value to estimate the % of sample that would have been >2 mm in the 10 g subsample. For particle sizes occurring between 2 to 5 mm diameter (<10% of total particles, KOM, unpublished data), we assigned a midpoint size of 3.5 mm and included them in mean weighted particle size calculations.

We quantified the relative abundance of coarse woody debris (CWD) in each stream during April 2002 and March 2003 using a modified transect method (see Wallace and Benke 1984). We quantified all submerged CWD >2.5 cm in diameter and all CWD buried within the upper 10 cm of the substrate along 15 cross-stream transects per stream; individual transects were 1-m wide with adjacent transects being spaced longitudinally 5-m apart. Live woody material (i.e., roots) was abundant in our study



streams and appeared to be an important influence on channel structure (KOM, personal observations), so we also included all live material in CWD measurements. CWD data were converted to planar area ( $\text{m}^2$  of CWD per  $\text{m}^2$  of stream bed) by multiplying CWD diameter by length and then dividing this value by the area sampled within each transect.

Dissolved organic carbon.—We measured streamwater dissolved organic carbon (DOC) on 1 date every 2 mo from November 2001 to September 2002, with 1 grab sample collected per stream per date using a 60-mL syringe. The syringe was fitted with a 0.45  $\mu\text{m}$  HPLC Gelman Acrodisc® syringe filter and ~30 mL was filtered into a pre-acid washed polycarbonate bottle. We then shipped samples on ice to the Oak Ridge National Laboratory, Oak Ridge, Tennessee, where DOC was measured by high-temperature combustion using a Shimadzu Model 5000 TOC analyzer after acidification and purging to remove inorganic C.

### Data analysis

Preliminary analyses using a repeated-measures ANOVA to examine seasonal variation (spring, summer, winter), revealed no season effect for any of the parameters measured (KOM, unpublished data). Therefore we used average seasonal values in analyses. Preliminary analysis using simple correlation revealed no significant relationship between selected land cover/use variables. However, many physical and organic matter variables were interrelated (i.e., 1 variable potentially affecting another) so we used simple linear correlation to detect bivariate relationships between



dependant variables. We used multiple regression to determine predictive relationships  
250 between physical and organic matter and land use and natural landscape variables.  
We used Akaike's Information Criteria adjusted for sample size ( $AIC_c$ ) and used  
adjusted coefficients of determination ( $R^2_{adj}$ ) for model selection. The regression model  
with the smallest  $AIC_c$  value was considered the best model of the measured variation  
in the data; however, we also considered all models with  $<2$   $AIC_c$  units of the overall  
255 best model ( $\Delta AIC_c < 2$ ) to have substantial support (Burnham and Anderson 2002).  
Analysis of multicollinearity using variation inflation factors (VIFs) revealed no highly  
multicollinear land cover/use variables (i.e. all VIFs  $< 10$ , Myers 1990). We transformed  
% CWD and % BPOM (arcsine square root) and particle size data ( $\log_{10}$ ) prior to  
analysis to satisfy normality. All remaining variables were normally distributed and  
260 therefore received no transformation.

## Results

The proportion of bare ground and road cover in study catchments (%BGRD)  
ranged from ~2 to 15% (mean = 8%), whereas the proportion of nonforested land  
265 (%NF) ranged from ~6% in BC2 to 38% in SB2 (mean = 22%, Table 2). The % of the  
catchment containing sandy soils (%Sand) ranged from ~2% in BC2 to almost 100% in  
SB3 (mean = 29%). In general, catchments within military compartments associated  
with heavy-tracked vehicular training showed higher %BGRD and %NF (LPK, SB1,  
SB2, SB4, SB5) than compartments without such mechanized training (BC1, BC2, HB,  
270 KM1, KM2, SB3, Tables 1 and 2).



Not surprisingly, stream physical and organic matter variables often were intercorrelated (Table 3). For correlations involving % submerged CWD and % BPOM, we removed 1 stream (BC1) from analysis following preliminary diagnostics. This catchment showed an atypical flood plain (see below). The percent of submerged CWD was negatively correlated with flashiness. Dissolved organic carbon and %BPOM were positively correlated with % submerged CWD (Table 3). Streambed instability was negatively correlated with mean particle size (Table 3). Particle size also was negatively correlated with flashiness but positively so with % submerged CWD (Table 3).

Analysis of relationships between land use and stream variables indicated that %BGRD was the best single predictor for many physical and OM parameters (Table 4). Streambed instability was positively correlated with %NF ( $\beta_0 = 1.87$ ,  $\beta_{\%NF} = 0.08$ ,  $\underline{R}_{adj}^2 = 0.43$ , Figure 3), whereas stream flashiness was negatively correlated with storm magnitude (as  $\max[Q/Q_{base}]$ ) and %Sand ( $\beta_0 = -0.041$ ,  $\beta_{\max(Q/Q_{base})} = 0.010$ ,  $\beta_{\%sand} = 0.003$ ,  $\underline{R}_{adj}^2 = 0.74$ , Table 4). However, the univariate model containing %BGRD also explained a high amount of variation in stream flashiness and thus had support ( $\beta_0 = 0.067$ ,  $\beta_{\%BGRD} = 0.020$ ,  $\underline{R}_{adj}^2 = 0.54$ , Figure 4A). Sally Branch 4 (SB4) was the only stream with an undefined channel and thus could have been considered an outlier (i.e.,  $>2$  SD from mean recession constants for other streams). When we removed SB4 from the analysis, the best 2-factor model for stream flashiness consisted of %BGRD and %NF ( $\beta_0 = -0.049$ ,  $\beta_{\%BGRD} = -0.037$ ,  $\beta_{\%NF} = 0.003$ ,  $\underline{R}_{adj}^2 = 0.94$ ). However, the univariate model containing %BGRD also explained a large amount of variation in flashiness with the removal of SB4 from the analysis ( $\beta_0 = -0.014$ ,  $\beta_{\%BGRD} = -0.031$ ,



$R^2_{adj} = 0.85$ , Table 4). Mean substrate particle size was negatively correlated with

295 %BGRD ( $\beta_0 = -0.054$ ,  $\beta_{\%BGRD} = -0.015$ ,  $R^2_{adj} = 0.45$ , Figure 4B).

All 3 stream organic matter variables (CWD, BPOM, DOC) were inversely correlated with %BGRD (Figure 5). For submerged CWD and %BPOM 1 catchment (BC1) showed %BPOM and CWD amounts that were >2 SD the mean, which prompted us to exclude this site from regressions. This catchment had an unusually broad  
300 floodplain and a high riparian stand density, both of which likely increased streambed organic matter retention. The proportion of bare ground and unpaved road cover best explained % submerged CWD ( $\beta_0 = 0.358$ ,  $\beta_{\%BGRD} = -0.013$ ,  $R^2_{adj} = 0.79$ , Figure 5A), %BPOM ( $\beta_0 = 0.171$ ,  $\beta_{\%BGRD} = -0.007$ ,  $R^2_{adj} = 0.62$ , Figure 5B), and DOC ( $\beta_0 = 4.04$ ,  $\beta_{\%BGRD} = -0.132$ ,  $R^2_{adj} = 0.32$ , Figure 5C). A 2-variable model including %NF (negative  
305 correlation) and %Sand (positive) best explained the % of buried CWD ( $\beta_0 = 0.015$ ,  $\beta_{\%NF} = 0.0007$ ,  $\beta_{\%sand} = -0.0002$ ,  $R^2_{adj} = 0.34$ , Table 4).

## Discussion

Military installations often have repeated and high-magnitude, but localized  
310 perturbations associated with training, causing soil disturbance, increased erosion, and sedimentation in streams. At FBMI, heavy-tracked vehicle training (i.e., tank maneuvers), munitions impact areas, unpaved roads, controlled burning, and timber harvesting all contributed to terrestrial disturbance (USAIC 2001). Although our study was correlative and thus could not determine specific causal mechanisms of landscape  
315 change our results suggest that the proportion of a catchment denuded of vegetation and with exposed and constantly disturbed soil (%BGRD) was a key terrestrial influence



on stream geomorphology, hydrology, and organic matter state. In general, land use variables (e.g., proportion of bare ground and road cover, nonforested land) were far better predictors of instream physical and organic matter conditions than were natural geomorphic or topographic attributes (catchment size and soil type).

#### Influence of military land use on stream geomorphology and hydrology

Land use practices such as urbanization, agriculture, and forest harvest often deliver eroded soil to stream channels (Ryan 1991, Waters 1995, Sutherland et al. 2002). Increased sedimentation may increase streambed instability (i.e., increase threshold entrainment per unit discharge, Lorang and Hauer 2003) because of reduced shear resistance associated with recent deposition of unstable particles (Jain and Park 1989, Krone 1999). Our results at FBMI are consistent with these findings, in that streambeds draining catchments with a higher proportion of nonforested land were less stable than those draining less-disturbed, forested catchments. It must be noted, however, that grassland and shrubland in our nonforested category (%NF, Figure 2D) probably represented land recovering from not only contemporary military training and silviculture, but also from historical agriculture, a likely additional sediment source for streams (Trimble 1981, 1999). The FBMI landscape consisted of extensive agriculture prior to military purchase in the early 1940s (Kane and Keeton 1998, USAIC 2001), so a 'legacy' effect (sensu Harding et al. 1998) of sediment input from historical agriculture is plausible, with sediments eroded during the agricultural period continuing to migrate through ephemeral channels and into perennial streams. As with agriculture, use of heavy-tracked vehicles for military training is generally limited to low to moderate slopes



of the installation, although in most catchments active sediment entry into streams from upland sources appears to occur mainly from contemporary land use, especially from active roads or bare ground used for training (Figure 2A &C). However, presence of active sediment movement in some stream channels at FBMI with no apparent upland source in the catchment (KOM, personal observations) would suggest that streambed instability and substrate composition may result from a combination of contemporary military/forestry and historical agriculture land use.

Stream flashiness increased as a function of increasing proportion of bare ground and road cover in catchments. One likely explanation for this pattern was that low vegetative cover in highly disturbed catchments caused higher and more temporally variable runoff. In addition, bare ground may amplify sealing of soil surfaces (Assouline and Mualem 2002) and use of heavy-tracked vehicles at FBMI are known to compact soil and decrease rainfall infiltration (Goran et al. 1983, Garten et al. 2003). Taken together, the combination of increased soil sealing, and reduced evapotranspiration and infiltration, likely contributed to increased overland flow in high %BGRD catchments, thus increasing the magnitude of short-term variation in stream hydrographs during storm events (Rose and Peters 2001).

Average streambed particle size often decreases with increasing agriculture, silviculture, or urbanization within catchments (Nerbonne and Vondracek 2001, Walters et al. 2003), usually because eroded particles entering streams are disproportionately fine grained (Bilby et al. 1989). We observed a similar pattern at FBMI, where catchments with a higher proportion of %BGRD had smaller streambed particle sizes than less-disturbed catchments. Moreover, average particle size was negatively related



to streambed instability and flashiness (Table 3), with both hydrologic variables being higher in highly disturbed streams (Table 4). Disturbed streams generally contained a higher proportion of active channel incision and bank erosion (KOM, personal observations), which probably supplied substantial fine-grained sediment to the stream bottom. Further, input of eroded sediment from incised ephemeral channels as a result of historical agriculture (pre-1940s) likely also contributed sediment to perennial stream beds. Collectively, our results suggest that disturbance from land use at FBMI alters particle size distributions in the stream bed, apparently from a combination of erosional (i.e., a disproportionate terrestrial input of fines) and hydrologic (i.e., increased bank erosion from high flashiness) influences, resulting in disproportionately high input of fine particles in disturbed catchments.

#### Influence of military land use on stream organic matter

Low instream CWD in disturbed (high %BGRD and %NF) catchments (Table 4) was consistent with studies of CWD relationships with other land-use practices, including timber harvest (Harmon et al. 1986, Webster et al. 1992). In this context, military and silvicultural land uses are similar in that both remove or reduce vegetation, decrease the amount of organic matter within the catchment, and hence potentially reduce detrital inputs to streams. In addition to lower organic matter inputs however, flashier streams with less stable beds in more highly disturbed catchments may increase burial and/or export of CWD, which in tandem may reduce CWD abundance in the surficial substrate. However, the high correlation between % of buried CWD and %NF in our streams (albeit not with %BGRD, Table 4), suggests that CWD burial may



be coincident with increased CWD transport from disturbed stream reaches (see Bilby and Bison 1998). Transport of CWD is a function of CWD size with respect to the wetted and bankfull channel width and discharge (Harmon et al. 1986, Bilby and Bison 1998). Coarse woody debris pieces within our streams were much smaller (median diameter = 5 cm, length = 35 cm) than both average wetted and bankfull stream width (median = 170, 235 cm, respectively), so it is possible that considerable CWD export occurred from study reaches during storms. Further, live roots were important components of the CWD measure composing ~23% of CWD abundance in study streams (range 0–50%), and abundance of live roots also were significantly higher in the 5 least disturbed compared with the 5 most disturbed streams (two-tailed  $t$ -test,  $p = 0.005$ ). Live roots are typically stable and also may accumulate debris (Smock et al. 1989), which likely increased CWD in low-disturbance streams. Historic removal of CWD (“stream cleaning”) in agricultural catchments also may have influenced abundance of contemporary instream CWD. However, the majority of study catchments experienced high historic disturbance from agriculture (range 22–54%, mean = 39% of catchment was historic (1944) bare ground and field cover, KOM unpublished data), so it is likely all catchments experienced a relatively similar historic cleaning of CWD within channels.

Organic matter in small temperate-deciduous streams is derived primarily from allochthonous inputs (Cummins 1974, Mulholland 1997). Forested streams generally show higher BPOM than disturbed nonforested streams (Golladay 1997), mainly because reduced riparian vegetation may directly decrease BPOM. At FBMI, most streams have intact riparian vegetation and high canopy cover (>90%, KOM,



unpublished data), so low BPOM in disturbed stream beds was not likely produced from

410 lower allochthonous inputs. A more plausible reason for lower BPOM in disturbed catchments was because of lower instream BPOM-retention structures, particularly that of low CWD in the stream channel (Bilby 1981, Smock et al. 1989, Wallace et al. 1995).

In our study, relative abundance of submerged CWD decreased with increasing %BGRD (Table 4) and CWD and % BPOM were highly correlated (Table 3), suggesting

415 that land use may directly reduce instream CWD, which in turn reduces BPOM.

Furthermore, higher stream flashiness in more disturbed catchments (high %NF and %BGRD) may have exacerbated effects of low CWD on BPOM by increased BPOM transport during storms (see Smock 1990, Smock 1997).

We also found lower streamwater DOC concentrations in highly disturbed (high 420 %BGRD) catchments. This result is consistent with patterns observed in clearcut catchments, where deforestation reduces subsurface, litter leachate, throughfall, and instream DOC inputs (Meyer and Tate 1983). Primary sources of streamwater DOC either are allochthonous, including C from terrestrial OM, precipitation, and throughfall (McDowell and Likens 1988, Qualls et al. 1991, Michalzik et al. 2001), or

425 autochthonous, such as from algal or cyanobacterial exudates (Kaplan and Newbold 1982, Meyer and Tate 1983, Mulholland and Hill 1997). At FBMI, exposed soil in our %BGRD category generally contains low levels of labile OM (Garten et al. 2003), hence reduction of this C source from soil may have lowered streamwater DOC inputs. In addition, streams in highly disturbed catchments typically showed much lower benthic 430 algal biomass (as chlorophyll *a*) than low-disturbance streams (~2 vs ~6 µg/L in disturbed vs undisturbed streams, respectively, Stephanie A. Miller, Auburn University,



unpublished data), which also may have contributed to lower DOC concentrations in disturbed streams.

435 Implications of military land use to stream ecology

Dramatic modification of stream geomorphological and hydrological conditions and organic matter states attributable to military land use at FBMI may cause severe impairment to biotic communities. In particular, decreased streambed stability, increased stream flashiness, and decreased CWD may all impact stream communities  
440 by affecting habitat stability, persistence, and abundance, especially for benthic algae (Tett et al. 1978, Yamada and Nakamura 2002) and macroinvertebrates (Benke et al. 1984, Angradi 1999). Moreover, streams with low CWD often show smaller, shallower pools and less cover for fish (Angermeier and Karr 1984, Inoue and Nakano 1998).

The US Department of Defense (DoD) manages ~30 million acres of land at  
445 ~6000 locations (USDoD 2004), which can provide fruitful areas for long-term ecological research involving landscape disturbance on stream structure and function. There are several other military bases in the southeastern U.S. (Fort Stewart, Fort Mitchell, Fort Polk) for which our results may be directly applicable, however our results also may apply to non-military lands in the region where sediment is the main stressor to streams.  
450 Moreover, apart from the need to characterize long-neglected impacts of landscape disturbance on instream processes on military lands, perhaps the greatest value of these installations is 1) the vast amount of supporting data available, and 2) the consistent, well-documented patterns of land use attributable to military training. Long-



term studies can thus be conducted at installations to increase our understanding of the  
455 contemporary and historical influences on receiving streams.



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Table 1. Locations and characteristics of study streams at Fort Benning Military Installation, Georgia. Infantry/ranger military land use primarily consists of foot traffic and associated personnel transport vehicles, which results in a low degree of disturbance; heavy machinery consists primarily of tracked vehicle training (tanks) and results in a high degree of disturbance, and impact military land use are areas where live and dud munitions detonate and results in a high degree of disturbance. ND = no data.

| Stream                     | Abbreviation | UTM                   | Military land use           | Stream order | Drainage area (km <sup>2</sup> ) | Mean stream slope (%) | Average wetted width (m) | Average wetted depth (m) | Average bankfull depth (m) | % canopy cover (summer) |
|----------------------------|--------------|-----------------------|-----------------------------|--------------|----------------------------------|-----------------------|--------------------------|--------------------------|----------------------------|-------------------------|
| Bonham Tributary           | BC1          | 0710893N,<br>3588286E | Infantry/ ranger            | 2            | 2.10                             | 1.67                  | 1.34                     | 0.16                     | 0.45                       | 94.19                   |
| Bonham Tributary           | BC2          | 0710627N,<br>3588976E | Infantry/ ranger            | 2            | 0.75                             | 2.67                  | 0.99                     | 0.10                     | 0.41                       | 94.59                   |
| Hollis Branch              | HB           | 0717848N,<br>3583123E | Infantry/ ranger            | 2            | 2.15                             | 2.00                  | 1.98                     | 0.12                     | 0.43                       | 91.86                   |
| Kings Mill Creek Tributary | KM1          | 0720701N,<br>3600036E | Infantry/ ranger            | 2            | 3.69                             | 0.83                  | 2.17                     | 0.17                     | 0.25                       | 95.22                   |
| Kings Mill Creek Tributary | KM2          | 0719946N,<br>3599991E | Infantry/ ranger            | 3            | 2.31                             | 1.00                  | 1.66                     | 0.19                     | 0.36                       | 94.34                   |
| Lois Creek                 | LC           | 0715377N, 3597908E    | Infantry/ ranger/<br>impact | 2            | 3.32                             | 1.69                  | 1.41                     | 0.14                     | 0.43                       | 91.67                   |
| Little Pine Knot Tributary | LPK          | 0719223N,<br>3585421E | Heavy machinery             | 2            | 0.33                             | 5.10                  | 0.98                     | 0.07                     | 0.24                       | 96.23                   |
| Sally Branch Tributary     | SB1          | 0716349N,<br>3585850E | Heavy machinery             | 3            | 5.43                             | 1.33                  | 1.96                     | 0.10                     | 0.52                       | ND                      |
| Sally Branch Tributary     | SB2          | 0716808N,<br>3584787E | Heavy machinery             | 2            | 1.23                             | 2.31                  | 1.58                     | 0.09                     | 0.39                       | 93.33                   |
| Sally Branch Tributary     | SB3          | 0716673N,<br>3584684E | Infantry/ ranger            | 1            | 0.72                             | 1.00                  | 1.03                     | 0.07                     | 0.31                       | 95.04                   |
| Sally Branch Tributary     | SB4          | 0716005N,<br>3584889E | Heavy machinery             | 1            | 1.00                             | 1.33                  | 1.26                     | 0.07                     | 0.21                       | 88.50                   |
| Sally Branch Tributary     | SB5          | 0714935N,<br>3585249E | Heavy machinery             | 2            | 1.27                             | 1.67                  | 1.04                     | 0.06                     | 0.43                       | 95.16                   |



Table 2. Results of land use classification. Stream abbreviations defined in Table 1.

| Stream | % of<br>Catchment as<br>bare ground on<br>slopes >3% | % of<br>catchment as<br>unpaved<br>roads | % of<br>catchment as<br>trails | % of<br>catchment as<br>bare ground<br>and roads | % of<br>catchment<br>as<br>nonforests | % of<br>catchment<br>with sandy<br>soils | Number of<br>stream and road<br>crossings above<br>sampling site |
|--------|--|--|--------------------------------|--|---------------------------------------|--|--|
| BC1    | 7.1  | 2.7                                      | 0.6                            | 10.5   | 12.6                                  | 3.2                                      | 2  |
| BC2    | 0.3  | 1.5                                      | 1.3                            | 3.1  | 6.1                                   | 1.8                                      | 1  |
| HB     | 4.6  | 0.8                                      | 1.3                            | 6.6  | 26.6                                  | 54.1                                     | 3  |
| KM1    | 1.5  | 1.4                                      | 1.7                            | 4.6  | 14.4                                  | 35.7                                     | 0  |
| KM2    | 0.1  | 0.7                                      | 0.9                            | 1.8  | 13.1                                  | 17.8                                     | 0  |
| LC     | 2.8  | 0.0                                      | 0.9                            | 3.7  | 22.3                                  | 23.5                                     | 0  |
| LPK    | 9.7  | 0.0                                      | 1.6                            | 11.3   | 29.2                                  | 9.0                                      | 0  |
| SB1    | 5.8  | 1.1                                      | 1.5                            | 8.4  | 22.4                                  | 31.0                                     | 4  |
| SB2    | 4.8  | 1.8                                      | 1.6                            | 8.1  | 37.7                                  | 28.3                                     | 0  |
| SB3    | 9.1  | 0.8                                      | 0.6                            | 10.5   | 32.2                                  | 97.2                                     | 0  |
| SB4    | 10.6   | 1.7                                      | 1.4                            | 13.6   | 22.2                                  | 35.2                                     | 1  |
| SB5    | 10.4   | 2.9                                      | 1.3                            | 14.7   | 20.2                                  | 5.3                                      | 0  |



Table 3. Summary of univariate Pearson correlations among stream physical and organic matter variables observed within the 12 study streams. DOC = streamwater dissolved organic C concentration, % BPOM = proportion of substrate as benthic particulate organic matter, % CWD = proportion of stream bottom as coarse woody debris. \* $p \leq 0.10$ , \*\*  $p \leq 0.05$ .

|                       | DOC  | % BPOM              | Streambed instability | Mean particle size   | % submerged CWD     | % buried CWD       | Stream flashiness    |
|-----------------------|------|---------------------|-----------------------|----------------------|---------------------|--------------------|----------------------|
| DOC                   | 1.00 | 0.66** <sup>a</sup> | -0.52                 | -0.81** <sup>a</sup> | 0.59* <sup>a</sup>  | -0.46              | -0.56                |
| % BPOM                |      | 1.00                | -0.48 <sup>a</sup>    | 0.76** <sup>a</sup>  | 0.76** <sup>a</sup> | -0.05 <sup>a</sup> | -0.60 <sup>a</sup>   |
| Streambed instability |      |                     | 1.00                  | -0.82** <sup>a</sup> | -0.23 <sup>a</sup>  | 0.51               | 0.18                 |
| Mean particle size    |      |                     |                       | 1.00                 | 0.67* <sup>a</sup>  | -0.24 <sup>a</sup> | -0.65* <sup>a</sup>  |
| % submerged CWD       |      |                     |                       |                      | 1.00                | -0.25 <sup>a</sup> | -0.81** <sup>a</sup> |
| % buried CWD          |      |                     |                       |                      |                     | 1.00               | 0.34                 |
| Stream flashiness     |      |                     |                       |                      |                     |                    | 1.00                 |

<sup>a</sup>correlations excluding stream BC1 (see text).



Table 4. Selected results of multiple regression analyses to describe the relationship among land use and hydrological variables and stream response variables. Models for each variable are listed in increasing complexity. The regression with the lowest adjusted Akaike's Information Criterion ( $AIC_c$ ) was considered the best model, although models with a slight difference from the best model ( $\Delta AIC_c < 2$ ) also had substantial support (Burnham and Anderson 2002).

Max(Q/Qbase) was the maximum increase in discharge over base flow during a storm event, %BGRD, %NF, and %Sand were percentages of the catchment occurring as bare ground and road cover, nonforested, and sandy soil, respectively.

BC1= outlier stream, SSE = model sum of squares error, CWD = coarse woody debris, DOC = dissolved organic C concentration, % BPOM = percentage of benthic particulate organic matter in the stream bed.  $\underline{n}$  = 12 for buried CWD and DOC,  $\underline{n}$  = 11 for particle size, submerged CWD, and % BPOM, and  $\underline{n}$  = 9 for stream instability and flashiness. \*indicates model with  $\Delta AIC_c > 2$ , but had high amount of variation explained by the simple model.

| Parameter                          | Parameters in model       | Number<br>parameters<br>in model | $AIC_c$ | $\Delta AIC_c$ | SSE   | Adjusted<br>$R^2$ | p     |
|------------------------------------|---------------------------|----------------------------------|---------|----------------|-------|-------------------|-------|
| Stream flashiness<br>(all streams) | %BGRD                     | 2                                | -44.21  | 1.73           | 0.034 | 0.54              | 0.014 |
|                                    | max(Q/Qbase), %Sand       | 3                                | -45.94  | 0.00           | 0.016 | 0.74              | 0.007 |
|                                    | max(Q/Qbase), % NF, %Sand | 4                                | -44.83  | 1.11           | 0.008 | 0.84              | 0.006 |
| Stream flashiness<br>(without BC1) | %BGRD                     | 2                                | -46.44  | 3.39*          | 0.011 | 0.85              | 0.001 |
|                                    | %BGRD, %NF                | 3                                | -49.83  | 0.00           | 0.004 | 0.94              | 0.000 |
| Streambed instability              | %NF                       | 2                                | 1.24    | 0.00           | 5.3   | 0.43              | 0.033 |
|                                    | %NF, %Sand                | 3                                | 2.18    | 0.93           | 3.5   | 0.57              | 0.035 |
| Substrate size                     | %BGRD                     | 2                                | -56.30  | 0.00           | 0.040 | 0.45              | 0.014 |
| % submerged CWD                    | %BGRD                     | 2                                | -74.74  | 0.00           | 0.007 | 0.79              | 0.000 |
|                                    | %BGRD, % Sand             | 3                                | -73.15  | 1.59           | 0.006 | 0.81              | 0.001 |
| % buried CWD                       | %NF                       | 2                                | -114.54 | 0.08           | 0.001 | 0.19              | 0.086 |
|                                    | %NF, %Sand                | 3                                | -114.62 | 0.00           | 0.000 | 0.34              | 0.061 |
| % BPOM                             | %BGRD                     | 2                                | -79.98  | 0.00           | 0.005 | 0.62              | 0.002 |
| DOC                                | %BGRD                     | 2                                | -4.33   | 0.00           | 5.364 | 0.32              | 0.031 |
|                                    | %BGRD, % Sand             | 3                                | -4.26   | 0.07           | 3.976 | 0.44              | 0.029 |



Figure 1. Locations of study catchments (depicted by polygons) within Fort Benning Military Installation, GA. Dotted line in middle figure represents the Chattahoochee River, which separates Alabama (AL) and Georgia (GA). Numbers in the right figure identify watersheds on the same stream (e.g., 1 and 2 on the Bonham Creek represent Bonham Creek Tributaries 1 and 2, BC1 and BC2, respectively). Geographic coordinates and stream characteristics are given in Table 1.

Figure 2. Photographs illustrating typical heavy machinery training areas (A), trails (B), unpaved roads cover (C), and nonforested land (D) at the Fort Benning Military Installation, Georgia. Note the unstable, vegetation-poor soils in A and C, the poorly defined road cover in C, and the exposed (unvegetated) soil in D.

Figure 3. Relationship between streambed instability, calculated as the mean absolute change in bed height from January to July 2003, plotted against the % of nonforested land in the catchment. (Mean  $\pm$  1SE).

Figure 4. Stream flashiness (4-h recession constants) calculated as the regression slope of the LN(flow) for 4 h following peak flow as a function of the % of bare ground in a catchment (A) and mean stream substrate particle size (B) plotted against the % of bare ground and road cover in a catchment. Triangle indicates outlier catchments ( $>2$  SD below the mean), SB4 for recession constant and BC1 for particle size and pH, that were excluded from analyses. (Mean  $\pm$  1SE).



Figure 5. Relationship between % of catchment as bare ground and road cover and average submerged coarse woody debris (CWD, A), benthic particulate organic matter (BPOM, B), and baseflow streamwater dissolved organic carbon (DOC) concentration (C). The triangle indicates an outlier catchment BC1 ( $> 2$  SD) that was excluded from analyses. CWD and BPOM are the arcsine square root transformed data. Plotted points are individual streams. (Mean  $\pm$  1SE).



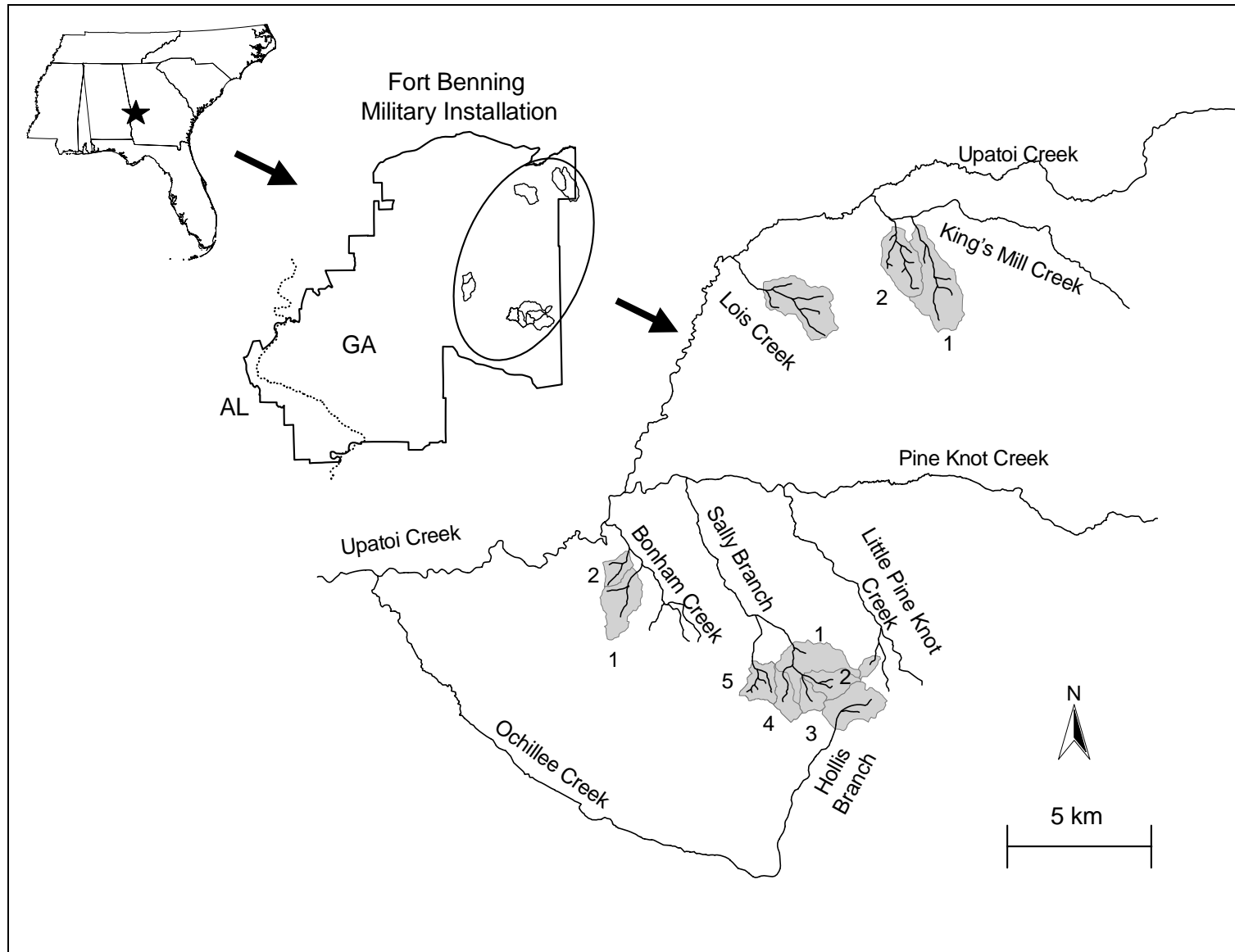


Figure 1.





Figure 2.



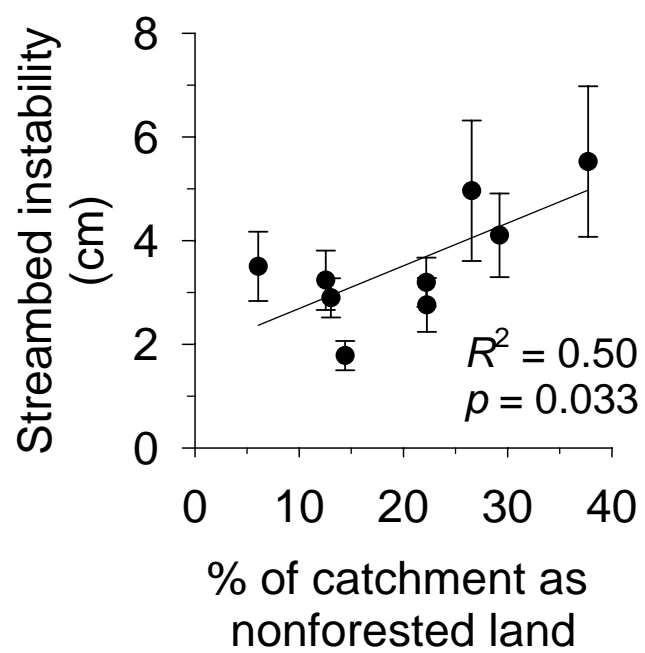


Figure 3.



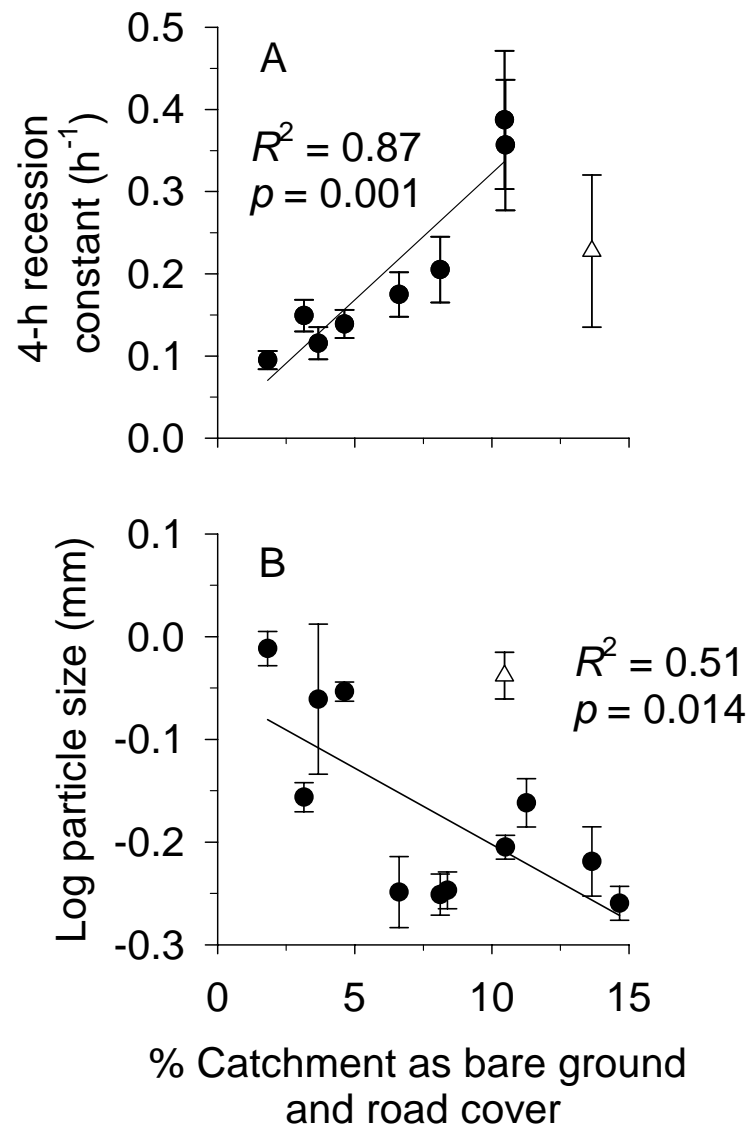


Figure 4.



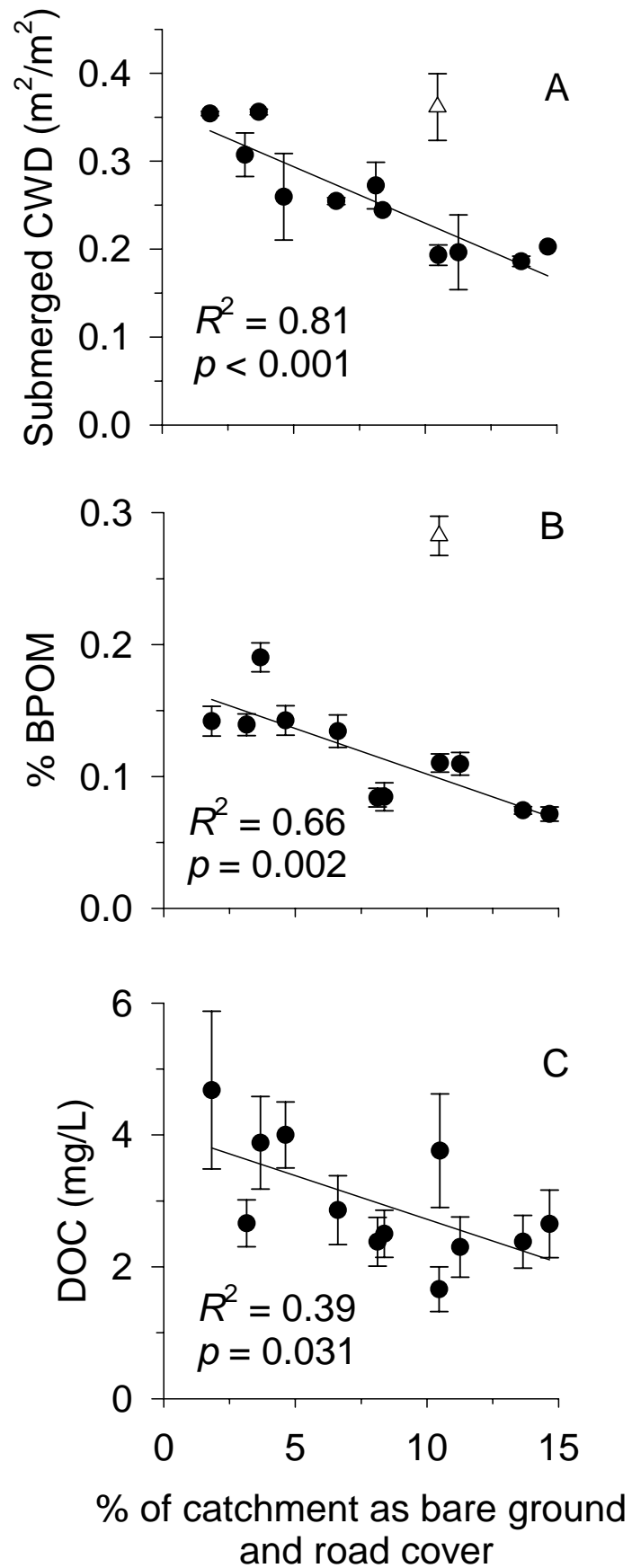


Figure 5.



Appendix B<sup>7</sup> of Indicators Final Report: Upland disturbance affects headwater stream  
nutrients and suspended sediments during baseflow and stormflow.

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**Abbreviations**

FBMI, Fort Benning Military Installation, Columbus, GA; BC1, BC2, tributaries of Bonham  
Creek; SB2, SB3, and SB4, tributaries of Sally Branch; KM1, KM2, tributaries of Kings Mill  
Creek; LPK, tributary of Little Pine Knot Creek; HB, Hollis Branch Creek; LC, Lois Creek;

20 TSS, total suspended sediments; ISS, inorganic suspended sediments; OSS, organic suspended  
sediments; DOC, dissolved organic carbon; SRP, soluble reactive phosphorus; DIN, dissolved  
inorganic nitrogen ( $\text{NH}_4^+ + \text{NO}_3^-$ ).



## ABSTRACT

Watershed characteristics determine the inputs of sediments and nutrients to streams. As a result, natural or anthropogenic disturbance of upland soil and vegetation may affect stream characteristics. The Fort Benning Military Installation (near Columbus, Georgia) experiences a wide range of upland disturbance intensities due to spatial variability in the intensity of military training. We used this gradient in disturbance intensity to investigate the effect of upland soil and vegetation disturbance on stream chemistry during baseflow and stormflow conditions.

During baseflow, total and inorganic suspended sediments were positively correlated with catchment disturbance intensity; [DOC] and [SRP] were negatively correlated with disturbance intensity; [NH<sub>4</sub><sup>+</sup>], [NO<sub>3</sub>], and [DIN] were not correlated with disturbance intensity. The increases in total, inorganic and suspended sediment concentrations during storms were much greater in the disturbed catchments. Disturbance had only a moderate effect on the increase in nutrient concentrations during storms. Adding soil descriptors did not significantly improve the regression models of concentration as a function of disturbance for any of the suspended sediment fractions, any nitrogen species, or pH. Catchment soil characteristics were significant predictors for [DOC], [SRP] and [Ca<sup>2+</sup>]. Despite the largely intact riparian zones of these headwater streams, upland soil and vegetation disturbance had clear effects on stream chemistry during baseflow and stormflow conditions.



## INTRODUCTION

Headwater streams are important sites of nutrient processing (Peterson et al., 2001) and are strongly influenced by their surrounding catchment, which is their primary source of organic material, nutrients, and sediment (Hynes, 1975; Vannote et al., 1980). The geology and land use of catchments affect the rate at which these substances are delivered to streams (Omernik, 1976; Richards et al., 1996). Because of the ubiquity of anthropogenic landscape modification (Hannah et al., 1994; Vitousek et al., 1997), understanding how streams function requires understanding the impacts of anthropogenic catchment disturbance.

Concentrations of nutrients and suspended sediments in headwater streams are controlled by a variety of anthropogenic and natural factors. Natural variation in inputs of substances to streams is observed due to seasonal differences in precipitation and evapotranspiration, and differences among catchments in geology, soil, and vegetation. However, in many catchments, anthropogenic modifications to the catchment have strong effects on the inputs of nutrients and sediments to streams. Urban and agricultural land use often increase the inputs of sediments and nutrients (*e.g.*, Allan et al., 1997; Strayer et al. 2003). Deforestation can dramatically increase sediment run off (*e.g.*, Gurtz et al., 1980; Kreutzweiser and Capell, 2001) and increase the export of nutrients, particularly nitrogen, to streams (*e.g.*, Likens et al., 1970; Harr and Fredriksen, 1988; Martin et al., 2000). The effects of catchment disturbance can be particularly strong during storm events (Webster et al., 1990).

The effects of disturbance within the riparian zone and the role of the riparian zone in mitigating some impacts of land use have been well-studied (*e.g.* Lowrance et al., 1984; Gregory et al., 1991; Osborne and Kovacic, 1993; Richards et al., 1996). Several intensive studies of the impacts of whole catchment deforestation have been conducted (Likens et al., 1970; Gurtz et al.,



1980; Webster et al., 1992) but less is known about how localized, intense soil and vegetation disturbance in upland areas affects streams with intact riparian zones.

Military reservations are well suited for studies of the impact of catchment disturbance on streams because of the broad range of disturbance intensities within these reservations and the proximity of highly disturbed and relatively undisturbed catchments. Military reservations are often regional islands of high quality habitat (Cohn, 1996), but, in areas dedicated to training exercises, intense soil and vegetation disturbance often occur (*e.g.*, Quist et al., 2003). As a result, these reservations contain a wide range of anthropogenic disturbance intensities within a small region of relatively homogenous soils, vegetation, and geology. In this study, we selected a set of streams spanning a gradient of disturbance at the Fort Benning Military Installation (FBMI) to investigate the impact of upland soil and vegetation disturbance on stream chemistry during baseflow and stormflow conditions.

## MATERIALS AND METHODS

### Study Site

Ten 2<sup>nd</sup> to 3<sup>rd</sup> order streams on the FBMI and within the Chattahoochee River Drainage of west-central Georgia were selected for study (Table 1; Fig. 1). Until purchased by the U.S military in 1918 and 1941-42 the land use was primarily row crop agriculture and pasture. Subsequently, the forest has regrown and, within the undisturbed areas on the base, land cover now consists primarily of oak-hickory-pine and southern mixed forest, with underlying sandy or sandy clay loam soils (Omernik, 1987). Dominant soil series found within catchments are Troup, Lakeland, Nankin, and Cowarts soils. The surficial geology of FBMI generally consists of Eutaw, Cusseta Sand, and Blufftown formations with a few, small areas of Tuscaloosa formation. In general, these formations are similar, and in some places, Eutaw can be difficult to



differentiate from Blufftown (Eargle, 1955) and Cussetta Sand (Veatch and Stephenson, 1911).

The dominant lithology of all four formations includes sand and clay. In addition, gravel may be present in Tuscaloosa (Veatch, 1909), marl in Blufftown, and lignite in Eutaw formations

(Hilgard, 1860; Veatch 1909). Certain areas of the reservation are frequently used for military

5 training involving infantry and mechanized heavy equipment (e.g. tracked vehicles such as tanks). As a result, some catchments have localized areas with high soil and vegetation disturbance resulting in bare ground, compaction of surface soils, and high rates of erosion (Fig. 2A-B). Other catchments have remained essentially undisturbed for the last 60-80 years.

The streams included in the study were typical low-gradient, sandy, Southeastern Plains

10 streams (Felley, 1992) with intact riparian canopies and mostly forested catchments. The riparian forest is almost entirely deciduous resulting in little shading of the streams during winter and spring. Leaf emergence usually occurs in late March or early April and leaf abscission usually occurs in early November. Thus, from late spring through early autumn the riparian trees provided a closed canopy and the streams were generally well shaded. Though precipitation

15 (averaging approximately 1 m annually) is distributed evenly throughout the year, stream discharge exhibits seasonal patterns. High rates of evapotranspiration during the growing season result in lower stream discharge in summer and autumn relative to winter and spring. The

following abbreviations are used to identify the streams included in this study: 2 tributaries of Bonham Creek, (BC1, BC2), 3 tributaries of Sally Branch (SB2, SB3, and SB4); 2 tributaries of

20 Kings Mill Creek (KM1, KM2); 1 tributary of Little Pine Knot Creek (LPK); Hollis Branch Creek (HB); and Lois Creek (LC) (Fig. 1).



### Disturbance Intensity

Disturbance intensity for each catchment was quantified by Maloney et al. (2005) and the methods are briefly reviewed here. Land use and basic soil types of the study catchments were quantified using geographic information system (GIS) datasets (streams, 1:24000, 1993; soils, 1:20000, 1998; and roads, 10 m, 1995), available digital orthophotography (1:5000, July 1999), digital elevation models (DEMs, 1:24000, grid size 10 m, 1993) and Landsat imagery (28.5 m, July and December 1999) provided by Fort Benning personnel and available on the U.S.

Department of Defense SERDP Environmental Management Program data repository

([www.sempdata.wes.army.mil](http://www.sempdata.wes.army.mil)). Catchment soils were characterized as the percentage of the catchment with soil in each of three categories: sand (Ailey loamy coarse sand and Lakeland sand soils), sandy clay loams (Nankin sandy clay loam soils) and loamy sands (dominated by Troup loamy sand, Cowarts and Ailey soils). Disturbance intensity was defined as the percent of unpaved road area and other areas of bare ground on slopes > 3% in each catchment. At Fort Benning, much of the bare ground was created by military training using tracked vehicles. Unpaved roads are largely roads and trails used by tracked military vehicles. The areas of soil and vegetation disturbance were located in upland areas away from the perennial streams, but most were hydrologically connected during storms via ephemeral drainages discharging to the perennial stream. Of the 249 Ft. Benning catchments, 245 have disturbance intensities between 0 to 17 % (K. Maloney, unpub. data). The 10 catchments included in this study spanned much of this disturbance gradient with disturbance intensities ranging from 1.8 – 13.7% (Table 1).



## Field Methods

Baseflow chemistry grab samples were taken from the thalweg with care to avoid suspending stream sediments. Samples were collected once or twice quarterly under baseflow conditions (defined as at least 2 days after peaks in stream discharge due to storms).

5 Stormflow chemistry samples were collected using ISCO autosamplers (Model 6700) equipped with water depth and ultrasonic flow velocity sensors (Model 710) and configured to collect a maximum of 24 samples during a storm. The sampler intake and depth/velocity sensor were positioned in the thalweg of each stream. The ISCO sampler was programmed to begin  
10 point. Samples were collected every 30 or 60 minutes (depending on the stream and season) for the duration of the period that stream depth remained elevated or until 24 samples were collected. Discharge during storms was calculated by the ISCO sampler using the water depth and velocity data recorded at 15-minute intervals and data describing stream channel shape (which was measured and entered during ISCO sampler deployment). Stormflow chemistry  
15 samples were collected across a number of storms and seasons for each stream. Stream discharge was also determined at the time of sample collection by salt dilution gauging using a continuous NaCl injection (for details see: Houser et al., 2005). Stream width was the average of wetted width measurements taken every 5 meters along 50 to 100 m reaches of each stream.

## 20 Chemical Analysis

Suspended sediment (total, inorganic and organic) concentrations were determined gravimetrically on samples filtered through pre-combusted and tared glass fiber filters (Whatman GFF) and with total suspended sediment mass ( $TSS_{mass}$ ) determined using an electronic balance



after drying (80 °C). Sample volumes of 100 to 700 mL were filtered depending on sample concentrations. Inorganic suspended sediment mass ( $ISS_{\text{mass}}$ ) was determined by combusting (500 °C for 12 hours), rewetting to restore the water of hydration, drying and reweighing.

Organic suspended sediment mass ( $OSS_{\text{mass}}$ ) was determined by difference ( $OSS_{\text{mass}} = TSS_{\text{mass}} - ISS_{\text{mass}}$ ).

Samples for dissolved N and P analyses were frozen after filtration until analysis.

Concentrations of soluble reactive phosphorus ([SRP]) were determined by the ascorbic acid-molybdenum blue method (APHA 1992) using a 10-cm spectrophotometer cell to achieve low detection limits. Ammonium concentration [ $NH_4^+$ ] was determined by phenate colorimetry

(American Public Health Association, APHA 1992) and [ $NO_2^- + NO_3^-$ ] by Cu-Cd reduction followed by azo dye colorimetry (APHA 1992), both using a Bran Lubbe autoanalyzer

(TRAACS Model 800 or AA3). Because stream water was always relatively high in dissolved oxygen concentration ( $> 6 \text{ mg L}^{-1}$ ) and because spot checks revealed minimal [ $NO_2^-$ ] ( $< 2 \text{ } \mu\text{g N L}^{-1}$ ), hereafter we refer to measurements of [ $NO_2^- + NO_3^-$ ] as [ $NO_3^-$ ]. Concentration of dissolved

organic carbon ([DOC]) was determined by high temperature combustion using a Shimadzu Model 5000 TOC analyzer. Concentration of total dissolved nitrogen ([TDN]) was determined

by high temperature combustion (Shimadzu TNM-1). Concentration of dissolved organic nitrogen ([DON]) were determined as the difference between [TDN] and total inorganic N

concentration ( $[NH_4^+] + [NO_3^-]$ ). Samples for major cations were acidified after filtration (0.5%

$HNO_3$ ) and [ $Ca^{2+}$ ] and [ $Mg^{2+}$ ] were determined by inductively coupled plasma emission

spectrometry. Samples for  $Cl^-$  and  $SO_4^{2-}$  analysis were refrigerated after filtration and analyzed by ion chromatography.



## Statistical Analysis

A number of statistical analyses were used to understand the factors affecting stream chemistry. Seasonal differences in stream concentration were analyzed using the Scheffé test for multiple comparisons (GLM procedure; SAS, 2000). Multiple regression was used to test for the effects of disturbance and soil characteristics on stream chemistry. The baseflow chemistry data was transformed (by logarithm or square root) where necessary to normalize residuals or stabilize variance. Spearman rank correlation was used to test for correlations among soil characteristics and disturbance. Because the stormflow chemistry data could not be transformed to produce approximately normal residuals and homogeneity of variance in regression against disturbance, spearman rank correlational analysis was used for analysis of the effects of disturbance and soil characteristics on storm chemistry.

## RESULTS

### Baseflow Chemistry: Seasonal Patterns

There were moderate seasonal differences in stream discharge and concentrations of suspended sediments, dissolved organic carbon and nutrients. Maximum stream discharge occurred in spring and minimum stream discharge occurred in summer and fall (Fig. 3A). Spring discharge was significantly higher than all other seasons. Differences in discharge among the other seasons were not significant. Minimum suspended sediment concentrations ([TSS], [OSS], and [ISS]) occurred in winter (Fig. 3B-D), which was a period of intermediate discharge in these streams. [TSS] and [ISS] were significantly higher in summer, the period of minimum discharge, than in other seasons. There was not a significant difference in [OSS] among spring, summer, and fall. Similar to [OSS], minimum [DOC] occurred in winter and there were no



significant differences among spring, summer and fall [DOC] (Fig. 4A). Summer [SRP] was significantly higher than spring and fall, and winter [SRP] was not significantly different from any other season (Fig. 4B). There were no significant seasonal patterns in  $[\text{NH}_4^+]$ ,  $[\text{NO}_3^-]$ , conductivity, or pH (Fig. 4C-F).

## 5 **Baseflow Chemistry: Effects of Disturbance**

Among stream differences in chemistry were strongly influenced by catchment disturbance intensity. Disturbance was a significant predictor of total suspended sediment (TSS) concentrations during baseflow (Table 2). Mean [TSS] ranged from approximately  $4.5 \text{ mg L}^{-1}$  in the least disturbed catchments to as high as  $10.5 \text{ mg L}^{-1}$  in the most disturbed catchments (Fig. 5A). In low disturbance catchments (disturbance intensity  $< 7\%$  catchment area), mean [TSS] was less than  $6 \text{ mg L}^{-1}$ . In high disturbance catchments (disturbance intensity  $> 7\%$  catchment area) mean [TSS] was generally greater than  $6 \text{ mg L}^{-1}$  and was more variable among streams. BC1 was an exception to this pattern. BC1 drained a catchment that had a notably broader, flatter floodplain than most of the other study catchments and this broad floodplain may have provided greater protection from the impacts of disturbance. BC1 was included in all figures, but was omitted from all statistical analyses.

The pattern in baseflow [ISS] across the disturbance gradient was similar to that seen for [TSS]. There was a significant increase in [ISS] with increasing disturbance intensity (Table 2), and BC1 was an outlier (Fig. 5B). At sites with disturbance intensities less than 7% of the catchment, mean [ISS] ranged from  $2.0 \pm 0.5$  to  $3.8 \pm 0.9 \text{ mg L}^{-1}$ , whereas streams in catchments with disturbance intensities greater than 7%, mean [ISS] ranged from  $5.6 \pm 0.9$  to  $7.3 \pm 1.3 \text{ mg L}^{-1}$ . As was the case with [TSS], there was increased variability in [ISS] among



streams with increasing disturbance intensity. Mean [OSS] ranged from 1.1 +/- 0.3 to 4.0 +/- 0.7 mg L<sup>-1</sup> and there was not a significant correlation with disturbance intensity (Fig. 5C).

The effect of disturbance intensity on dissolved nutrients varied among constituents.

[NO<sub>3</sub><sup>-</sup>] and [NH<sub>4</sub><sup>+</sup>] did not increase significantly with disturbance intensity (Fig. 6A-B).

5 However, 3 streams with moderately high disturbance intensities had the highest [NO<sub>3</sub><sup>-</sup>] (Fig. 6A) and the stream with the highest disturbance intensity had the highest [NH<sub>4</sub><sup>+</sup>] (Fig. 6B).

There was a marginally significant ( $p=0.06$ ,  $R^2=0.4$ ) increase in [DIN] with increasing disturbance due to the uniformly low concentrations of inorganic N in the streams with low disturbance intensities (Fig. 6C; Table 2). [SRP] declined significantly from 6.2 +/- 0.7 mg L<sup>-1</sup> in the least disturbed catchment to 2.6 +/- 0.3 mg L<sup>-1</sup> in the most disturbed catchment (Fig. 7A).  
 10 [DOC] declined significantly from 4.1 +/- 0.7 mg L<sup>-1</sup> in the least disturbed catchment to 1.5 +/- 0.2 mg L<sup>-1</sup> in the most disturbed catchment (Fig. 7B; Table 2).

There were significant increases in baseflow pH and [Ca<sup>2+</sup>] with increasing disturbance intensity (Fig. 8A, B; Table 2) and there was a significant correlation between pH and [Ca<sup>2+</sup>]  
 15 (Fig. 8C; Table 2). Although [DOC] was negatively correlated with disturbance, it was not correlated with pH and its inclusion in a multiple regression model did not improve the prediction of pH. As was seen for suspended sediments, BC1 was an outlier in the relationship between pH and disturbance exhibiting pH levels similar to streams in lower disturbance catchments.

20 [Si] decreased significantly as disturbance intensity increased (Table 2). The three least disturbed catchments exhibited [Si] greater than 4 mg L<sup>-1</sup>, whereas the more disturbed catchments exhibited [Si] less than 3.7 mg L<sup>-1</sup> (Fig. 9A). There was no significant relationship between disturbance intensity and conductivity, [Cl<sup>-</sup>], or [SO<sub>4</sub><sup>-2</sup>] (Fig. 9B-D).



Stepwise regression analysis was used to investigate whether adding soil descriptors significantly improved the regression models for the above stream chemistry constituents or affected the proportion of variability explained by disturbance (Table 2). Disturbance was not significantly correlated with soil description (% sand:  $r=0.4$ ,  $p=0.3$ ; % loamy sand:  $r=-0.5$ ,  $p=0.17$ ; % sandy clay loam:  $r=0.4$ ,  $p=0.3$ , spearman rank correlation). Most catchments were dominated by two soil categories: sandy soils or loamy sands. Only SB4 and SB2 had greater than 4% sandy clay loam. Adding soil descriptors to the analysis did not significantly improve the regression models for any of the suspended sediment fractions, any nitrogen species, or pH. Catchment soil type was a significant predictor for [DOC], [SRP] and  $[Ca^{2+}]$ . However, for [DOC] and [SRP], disturbance explained much more variance than did soil type (Table 2). Percent loamy sand and % sandy clay loam both were significant predictors of  $[Ca^{2+}]$  and when these variables were included in the regression model, disturbance was not significant.

### **Concentration-discharge Relationships During Storm Events**

Discharge, suspended sediment, and solute concentration patterns during a storm in January 2003 are presented in Figure 10 for two streams, one draining a relatively low disturbance catchment and the other draining a relatively high disturbance catchment. These patterns are generally typical of other storms as well. With the exception of  $[Cl^-]$ , all concentrations increased during storms, relative to pre-storm values. The increases in [TSS], [ISS], [DOC], and  $[NO_3^-]$  were distinct and were larger in the more highly disturbed stream. In contrast, the patterns for [SRP] and  $[NH_4^+]$  were variable and concentrations were at times lower than pre-storm values.  $[SO_4^{2-}]$  declined initially and then increased as discharge peaked and declined.



Differences between concentrations on the rising and falling limb of a storm hydrograph can provide information concerning the sources of various dissolved and particulate substances.

Evans and Davies (1998) proposed the following model: 1) At baseflow stream concentrations reflect groundwater concentrations, 2) On the rising limb of the hydrograph, stream

5 concentrations reflect surface event water and mobilization from nearby sources, and 3) On the falling limb, stream concentrations reflect catchment soilwater. A clockwise hysteresis in the plot of concentration vs. discharge occurs when solute concentrations are higher on the rising limb of the storm hydrograph (e.g., Fig. 11A), and suggests that solute concentrations in surface event water exceed soilwater concentrations. An anticlockwise hysteresis occurs when solute  
10 concentrations are higher on the falling limb of the storm hydrograph (e.g., Fig. 11B), and suggests that solute concentrations in soilwater exceeds surface event water. Indeterminate shapes occur when neither limb of the hydrograph exhibits consistently higher solute concentrations. These inferences are generalities as there is often considerable variability in the shape of these plots among storm events in any given stream (Johnson and East, 1982; McDiffett  
15 et al., 1989), but they remain informative.

There was no apparent effect of disturbance on the shape of the concentration-discharge (C-Q) plots in this study, but differences among stream chemistry constituents were observed (Table 3). TSS exhibited higher concentrations on the rising limb of the hydrograph (clockwise) in 21 of the 22 storms where an unambiguous shape was observed in the concentration vs.  
20 discharge plot (Fig. 11A).  $[\text{SO}_4^{2-}]$  exhibited the opposite pattern with concentrations being higher on the descending limb of the hydrograph (anti-clockwise) in 18 of the 19 storms where an unambiguous shape was observed (Fig. 11B). DOC data is more limited, but showed higher concentrations on the falling limb of the hydrograph (anticlockwise) in 9 of the 11 storms where



an unambiguous shape was seen.  $\text{NO}_3^-$  exhibited higher concentrations on the rising limb of the hydrograph in 17 of the 20 storms where a clear pattern was observed (Fig. 11C). [SRP] and  $[\text{NH}_4^+]$  rarely showed an obvious pattern in C-Q diagrams. [SRP] was indeterminate in 26 out of 32 storms and  $[\text{NH}_4^+]$  was indeterminate in 18 out of 32 and evenly divided between clockwise and anti-clockwise for the remainder.

### Stormflow Chemistry: Effects of Disturbance

The impact of disturbance on individual stream chemistry parameters during storm events was evaluated using the maximum change in concentration during storms (i.e. the difference between baseflow concentration measured prior to the storm and maximum concentration observed during a storm). In all cases where there were significant changes in concentration during storms, the concentration increased.

The increase in [TSS], [ISS] and [OSS] during storm events was positively correlated with catchment disturbance. In catchments with a disturbance intensity of  $< 7\%$ , the mean maximum change in [TSS] ranged from 57 to  $300 \text{ mg L}^{-1}$  (Fig. 11A). In catchments with a disturbance intensity  $> 7\%$ , mean maximum change in [TSS] ranged from 847 to  $1881 \text{ mg L}^{-1}$ . In contrast to the patterns under baseflow conditions, BC1 did not appear to be an outlier in the relationship between storm TSS or ISS concentration increase and disturbance intensity. The variability in maximum change in [TSS] among storms within streams was greater for streams with disturbance intensities  $> 7\%$  than in streams with lower disturbance intensities.

The pattern in [TSS] appeared to be driven by the ISS fraction of suspended sediments which showed essentially the same pattern as TSS. In catchments with a disturbance intensity of  $< 7\%$ , the mean maximum change in [ISS] ranged from 38 to  $255 \text{ mg L}^{-1}$  (Fig. 11B). In catchments with a disturbance intensity  $> 7\%$ , mean maximum change in ISS ranged from 707 to



1378 mg L<sup>-1</sup>. Again, the variability in maximum change in [ISS] among storms was considerably greater in streams draining the more highly disturbed catchments.

The mean maximum change in [OSS] during storm events was also positively correlated with catchment disturbance (Fig. 11C). However, the change in [OSS] was small compared to the change in [ISS] reflecting the fact that most of the suspended sediments in transport during storms were inorganic. Unlike [TSS] and [ISS], [OSS] did not show an obvious break point between streams in catchments above and below 7% catchment disturbance. The stream in the second most disturbed catchment exhibited a much higher mean maximum change in [OSS] than any other stream.

The effect of disturbance on stormflow nutrients varied among constituents. Mean maximum change in [SRP] increased significantly with disturbance (Fig. 13A). The increase in mean maximum change in [NO<sub>3</sub><sup>-</sup>] with disturbance intensity was only marginally significant (Fig. 13B), and the variability within each site was high. Maximum change in [SRP] and [NO<sub>3</sub><sup>-</sup>] did not show the breakpoint near 7% disturbance intensity that was seen for maximum change in [TSS] or [ISS]. The maximum change observed in storms for [DOC], [NH<sub>4</sub><sup>+</sup>] and [DIN] did not change significantly across the disturbance gradient.

The maximum increases in concentrations during storms correlated with catchment soil characteristics only for [DOC] and [NO<sub>3</sub><sup>-</sup>]. Change in [NO<sub>3</sub><sup>-</sup>] was negatively correlated ( $r = -0.67$ ,  $p=0.05$ ) with % loamy sand and exhibited a marginally significant correlation with % sandy clay loam ( $r=0.63$ ,  $p=0.06$ ). [DOC] was positively correlated with % sand ( $r=0.68$ ,  $p = 0.04$ ) and negatively correlated with % loamy sand ( $r=-0.68$   $p=0.04$ ).



## DISCUSSION

We studied a specific disturbance type (upland soil and vegetation disturbance by military training activities that results in bare ground and highly disturbed soils) in relative isolation of other disturbance factors. In contrast to whole catchment deforestation studies, we examined the impact of a moderate range of disturbance intensity (1.8 to 13.7 % of the catchment) in catchments with largely undisturbed riparian zones. However, the disturbance intensity in the areas we defined as disturbed were likely more severe than is typical in most types of deforestation (*e.g.*, timber harvest). Disturbance intensity affected stream chemistry and the magnitude of the disturbance effect varied widely among constituents. Variation in catchment soil characteristics were limited among our study catchments and had only moderate effects on a few stream constituents.

The strongest stream response to increased upland soil and vegetation disturbance intensity was an increase in suspended sediment concentration during baseflow and stormflow conditions (Figs. 5, 12). The magnitude of the increases in [TSS] and [ISS] during storms and their variability increased with disturbance intensity, and there appears to be a distinct difference between catchments with disturbance intensities  $< 7\%$  compared to those with disturbance intensities  $> 7\%$ . In streams in low disturbance catchments, maximum changes in [TSS] and [ISS] during storms were low regardless of magnitude of storm. In more highly disturbed catchments, maximum changes in [TSS] and [ISS] were much more variable, most likely due to effects of variations in storm size and antecedent conditions on suspended sediment concentrations in these catchments. However, normalizing storm TSS and ISS concentration increases to peak storm discharge did not improve the relationships with disturbance intensity.

These results agree with others who have observed increased suspended sediment



concentrations in response to deforestation and logging road construction (*e.g.*, Likens et al., 1970; Gurtz et al., 1980; Harr and Fredriksen, 1988; Kreutzweiser and Capell, 2001; MacDonald et al., 2003). However, our results differ from studies that have shown that riparian buffers can protect streams from suspended sediment inputs (*e.g.*, Martin et al., 2000; MacDonald et al., 2003). We found that once highly erodable soils are intensively disturbed, the eroded particles were effectively transported to the stream through the formation of ephemeral stream channels that connect the upland disturbed sites to the perennial stream channel. Once in stream channels and floodplains, redistribution and transport of catchment soil often continues for many years (Brown and Kyrgier, 1971). Swift (1988) found that 80% of the eroded soil from a forest clearcut remained in the stream channel after 2.5 yrs. In the current study, the stream sediments in the disturbed catchments are easily resuspended and higher suspended sediment concentrations are observed even during baseflow conditions. Only at our study site with the broadest riparian floodplain (BC1) did we observe an apparent riparian buffer effect on suspended sediment concentrations and this appeared to be confined to baseflow conditions only.

In addition to increases in total suspended sediment concentrations, catchment disturbance often causes an increase in the inorganic fraction of suspended sediments (*e.g.*, Likens et al., 1970; Bormann et al., 1974; Webster et al., 1992). In the current study, the increase in suspended sediment concentrations with disturbance was dominated by the inorganic fraction. There was not a significant increase in baseflow organic suspended sediments with disturbance intensity. The clay and sand dominated soils in these catchments do not appear to be an important source of organic material.

There was a strong negative correlation between [DOC] and disturbance intensity (Fig. 7B). There are several possible explanations for this pattern. Disturbed sites have reduced



organic matter content in the upland catchment soils (Garten et al., 2003) perhaps indicating a lower reservoir of leachable organic matter for output to the streams in disturbed catchments. In-stream DOC sources can be important in determining stream [DOC] (Bormann et al., 1974; Schiff et al., 1990; Webster et al., 1992). The potential instream sources of DOC, such as

5 benthic organic matter (BOM) and coarse woody debris (CWD), are reduced in our disturbed streams (Maloney et al., 2005). A third possible cause of lower [DOC] in the more highly disturbed catchments is the high concentration of inorganic sediments in the disturbed streams that may result in increased DOC adsorption in these streams. Clays are known to have high sorption potential for DOC (Jardine et al., 1989) and the sediments of the more highly disturbed

10 streams generally have higher content of clay-size particles (Maloney et al., 2005). Other studies have found forest cutting to result in an increase in DOC export initially (Hobbie and Likens, 1973); however, after this initial increase, DOC exports have been shown to fall to intensities below those found in undisturbed catchments (Meyer and Tate, 1983). The initial disturbance of the current study watersheds occurred prior to the current study. An initial increase in [DOC]

15 may have occurred that was not observed by the current study.

There was a strong negative correlation between [SRP] and disturbance intensity (Fig. 7A). This finding contrasts with others who have found no relationship between dissolved phosphorus and land use or ecosystem disturbance (Prairie and Kalff, 1988; Huryn et al., 2002). SRP is strongly adsorbed to inorganic particles, particularly clays (Meyer, 1979; Meyer and

20 Likens, 1979; Hill, 1982; Wood et al., 1984; Klotz, 1985), and the higher concentrations of inorganic suspended and streambed sediments in the more disturbed streams may result in higher rates of SRP adsorption resulting in lower streamwater [SRP]. An alternative mechanism is reduced instream sources of SRP (rem mineralization of organic P) in highly disturbed streams due



to reduced abundance of CWD and BOM. However, previous work suggests that the opposite trend in [SRP] should result from decreases in CWD and BOM because these materials can be major sites of phosphate uptake (Munn and Meyer, 1990). A third possible cause of reduced [SRP] is reduced SRP inputs from highly disturbed catchments. The reduced abundance of organic matter in the soils in disturbed catchments (Garten et al., 2003) may result in reduced leaching of SRP to soil water and ultimately to the stream.

There were no strong patterns in baseflow concentrations for nitrogen species across the gradient of catchment disturbance (Fig. 6). This contrasts with studies that have shown [ $\text{NO}_3^-$ ] to exhibit a strong response to catchment deforestation or other disturbances (*e.g.*, Likens et al., 1970; Bormann et al., 1974; Harr and Fredriksen, 1988; Martin et al., 2000; Huryn et al., 2002; Yeakley et al. 2003). The current study examined the effects of chronic upland disturbance. It is possible that initial increases in [ $\text{NO}_3^-$ ] would have occurred prior to the current study. Retention and denitrification in the intact riparian zone of these streams may also reduce the export of  $\text{NO}_3^-$  to the streams such that differences in transport from the upland areas to the riparian are not reflected in stream concentrations (Peterjohn and Correll, 1984; Hill, 1996; Hill et al., 2000).

The effect of catchment disturbance on storm increases in nutrient concentrations was considerably weaker than was observed for suspended sediments (Figs. 12 and 13). The increase in [SRP] during storms was significantly higher in disturbed catchment streams, and the relationship between increase in [ $\text{NO}_3^-$ ] during storms and disturbance intensity was marginally significant. Further, the increase in storm [SRP] with increasing disturbance intensity was opposite the effect disturbance intensity had on baseflow [SRP]. The reasons for the difference in disturbance effects on [SRP] patterns during baseflow and stormflow conditions may be related to disturbance effects on catchment soils and inorganic sediments in the streams. Under



baseflow conditions, soils and streambed sediments may be a sink for SRP via sorption to inorganic particles. However, erosion and suspension of inorganic sediments in runoff during storms may result in net desorption of P. For  $\text{NO}_3^-$ , larger storm increases in concentration with higher intensities of catchment disturbance is consistent with other studies showing higher  $[\text{NO}_3^-]$  concentrations and exports with disturbance (*e.g.*, Likens et al., 1970). Because  $[\text{NO}_3^-]$  in rainfall is likely much higher than  $[\text{NO}_3^-]$  in streamwater in the current study streams, the greater storm concentration increases suggest lower rates of retention of atmospheric N deposition in the more disturbed catchments, perhaps due to lower retention by vegetation and soil microbes in the denuded areas of these catchments.

There was a strong positive correlation between pH and disturbance (Fig. 8), but the mechanisms that caused this relationship are not clear. Catchment deforestation has been found to significantly increase  $\text{H}^+$  export, resulting in a decrease in stream pH (Likens et al., 1970; Martin et al., 2000; Yeakley et al., 2003). We see the opposite pattern here among catchments with an increase in pH with disturbance intensity. Lower pH in the undisturbed sites may be partly caused by the higher [DOC], but the relationship between [DOC] and pH was not significant. pH was not significantly correlated with soil characteristics. However, pH was significantly correlated with  $[\text{Ca}^{2+}]$  which was significantly correlated with catchment soil characteristics. Thus, the strong positive correlation between pH and disturbance may be the result of both disturbance effects and soil characteristics. Surficial geology appeared to be unimportant in the pattern of pH concentration among streams; there was no correlation between the fraction of a catchment underlain by the Blufftown formation, the only calcareous formation that occurs in our catchment, and pH.



There was no apparent effect of catchment disturbance on baseflow  $[\text{Cl}^-]$ ,  $[\text{SO}_4^{2-}]$ , or conductivity (Fig. 9B-D). This contrasts with the findings of a number of catchment deforestation studies.  $[\text{SO}_4^{2-}]$  generally decreases in response to catchment deforestation (Likens et al., 1970; Martin et al., 2000; Yeakley et al., 2003), whereas conductivity and  $[\text{Cl}^-]$  have been shown to increase (Likens et al., 1970).  $[\text{Si}]$  significantly decreased with disturbance intensity in our study (Fig. 9A), suggesting that there may be differences in subsurface flow path or decreases in soil-water residence time that resulted in reduced  $[\text{Si}]$  weathering in the disturbed catchments.

### Discharge vs. Concentration During Storm Events

We observed distinct shapes in the concentration vs. discharge plots for some stream chemistry constituents (Fig. 11; Table 3), but there were no apparent effects of disturbance intensity on these shapes. Suspended sediment concentrations were almost always higher on the rising limb of the hydrograph and peak suspended sediment concentrations usually preceded peak discharge (Figs. 10B, 10G, 11A). This suggests that surface runoff and mobilization from nearby areas is the dominant source of suspended sediments. Although the disturbed upland areas are likely the ultimate source of the excess suspended sediment in these streams, sediment that has been transported to the ephemeral stream channels or into the stream itself appears to be responsible for the initial large increase during most storms.

$[\text{DOC}]$  always increased during storm events, and  $[\text{DOC}]$  was generally higher on the falling limb of the storm hydrograph (anticlockwise; Figs. 10C, 10H), suggesting that soilwater is an important source of  $[\text{DOC}]$  during storms. This further suggests that the reduced organic material in the soils of disturbed catchments is at least partly responsible for the significant



decline in [DOC] in these streams as disturbance intensity increases. Others have observed higher [DOC] on the rising limb of the hydrograph in mountain streams (McDowell and Fisher, 1976; Meyer and Tate, 1983), suggesting that there are differences in sources of [DOC] among upland and lowland streams. Interpretation of the concentration vs. discharge patterns for [DOC] in the current study may be complicated by the possible effects of higher DOC adsorption during the rising limb of the hydrograph due to high inorganic suspended sediment concentrations.

Stormflow  $[\text{NO}_3^-]$  was generally higher on the rising limb of the hydrograph (Figs. 10D, 10I, 11C) suggesting that surface runoff (transport of rainfall  $\text{NO}_3^-$  or leaching of accumulated  $\text{NO}_3^-$  from surface soils), or riparian zone soils are major sources. A similar pattern was observed by McDiffett et al. (1989), but Buffam et al. (2001) observed the opposite, anticlockwise pattern with higher  $[\text{NO}_3^-]$  occurring on the falling limb of the hydrograph. Buffam et al. attributed this pattern to the  $\text{NO}_3^-$  build up in unsaturated soils throughout the catchment which was flushed to streams during the storm.

$[\text{SRP}]$  and  $[\text{NH}_4^+]$  were highly variable during storms and did not exhibit consistent differences between concentrations on the rising and falling limb of the hydrograph (Figs. 10C, 10H). As has been observed in other studies (Evans and Davies, 1998),  $[\text{SO}_4^{2-}]$  was consistently higher on the falling limb of the hydrograph suggesting a catchment soilwater source (Figs. 10E, 10J, and 11B).

## CONCLUSIONS

Upland soil and vegetation disturbance had clear effects on suspended sediments, nutrients and other aspects of stream chemistry. A relatively simple disturbance metric, the proportion of a catchment composed of bare ground on slopes greater than 3% and unpaved



roads, explained significant proportions of the variability for a group of stream chemistry parameters. The dominant effect of military training disturbance to upland vegetation and soils on streams was a large increase in inorganic suspended sediment transport. The increase was particularly evident during storms, but a significant effect existed during baseflow conditions as well. The increase in inorganic suspended sediments may have contributed to some of the other chemistry patterns that were observed, reducing baseflow [SRP] and [DOC] due to adsorption. These results illustrate that in regions with highly erodable soils, upland soil and vegetation disturbance can affect streams despite intact riparian zones. Sediment transport in ephemeral stream channels appears to reduce the effectiveness of the riparian zones in sediment retention.

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Table 1. Physical characteristics of the study stream reaches. Width, depth, flow, and velocity values are means and standard deviations based on measurements made during one NaCl injection conducted each quarter from the summer of 2001 through the summer of 2003.

| Stream | Width (m)   | Mean<br>depth (m) | Flow<br>(L s <sup>-1</sup> ) | Velocity<br>(m s <sup>-1</sup> ) | Catchment<br>area<br>(ha) | Disturbance<br>intensity<br>(% catchment) |
|--------|-------------|-------------------|------------------------------|----------------------------------|---------------------------|---|
| KM2    | 1.64 (0.39) | 0.15 (0.10)       | 16.58 (19.55)                | 0.04 (0.03)                      | 231                       | 1.8                                       |
| BC2    | 0.97 (0.08) | 0.11 (0.03)       | 4.85 (2.70)                  | 0.05 (0.02)                      | 74.9                      | 3.2                                       |
| LC     | 1.85 (0.20) | 0.12 (0.03)       | 16.64 (14.36)                | 0.07 (0.04)                      | 332                       | 3.7                                       |
| KM1    | 1.91 (0.22) | 0.13 (0.04)       | 25.61(13.67)                 | 0.1 (0.02)                       | 369                       | 4.6                                       |
| HB     | 1.77 (0.15) | 0.11 (0.03)       | 18.67 (14.9)                 | 0.09 (0.04)                      | 215                       | 6.6                                       |
| SB2    | 1.54 (0.14) | 0.06 (0.02)       | 14.65 (6.14)                 | 0.15 (0.03)                      | 123                       | 8.1                                       |
| BC1    | 1.33 (0.15) | 0.14 (0.03)       | 0.826 (3.91)                 | 0.04 (0.01)                      | 210                       | 10.5                                      |
| SB3    | 1.00 (0.13) | 0.05 (0.03)       | 6.15 (4.03)                  | 0.11 (0.03)                      | 71.7                      | 10.5                                      |
| LPK    | 0.77 (0.09) | 0.04 (0.02)       | 3.13 (1.45)                  | 0.10 (0.02)                      | 33.1                      | 11.3                                      |
| SB4    | 1.31 (0.47) | 0.04 (0.02)       | 6.60 (3.94)                  | 0.12 (0.03)                      | 100                       | 13.7                                      |



Table 2. Regression analysis results for baseflow concentrations of water chemistry parameters vs. disturbance level (Disturb.) and soil characteristics (percent sandy soil (per\_sand), percent loamy sand (per\_ls), and percent sandy clay loam (per\_scl)).

| Dependent var.                     | Independent var. | Estimate $\pm$ s.e.  | R <sup>2</sup> | p     |
|------------------------------------|------------------|----------------------|----------------|-------|
| sqrt(TSS)                          | Disturb          | 0.09 $\pm$ 0.02      | 0.70           | 0.005 |
| sqrt(OSS)                          | Disturb          |                      | ns             |       |
| log(ISS)                           | Disturb          | 0.08 $\pm$ 0.02      | 0.71           | 0.004 |
| sqrt(DOC)                          | Disturb          | -0.06 $\pm$ 0.0090   | 0.79           | 0.001 |
|                                    | Per_sand         | 0.003 $\pm$ 0.001    | 0.08           | 0.09  |
| sqrt(SRP)                          | Disturb          | -0.06 $\pm$ 0.02     | 0.75           | 0.008 |
|                                    | Per_ls           | 0.005 $\pm$ 0.002    | 0.11           | 0.07  |
| log(NH <sub>4</sub> <sup>+</sup> ) | Disturb          | 0.07 $\pm$ 0.04      | 0.32           | 0.1   |
| log(NO <sub>3</sub> <sup>-</sup> ) | Disturb          |                      | ns             |       |
| log(DIN)                           | Disturb          | 0.1 $\pm$ 0.05       | 0.4            | 0.06  |
| H <sup>+</sup>                     | Disturb          | -1.3E-6 $\pm$ 2.8E-7 | 0.75           | 0.003 |
| log(Ca <sup>2+</sup> )             | Disturb          |                      | ns             | ns    |
|                                    | per_ls           | -0.01 $\pm$ 0.004    | 0.56           | 0.01  |
|                                    | per_scl          | 0.04 $\pm$ 0.02      | 0.22           | 0.04  |
| log(conductivity)                  | Disturb          |                      | ns             |       |
| Cl <sup>-</sup>                    | Disturb          |                      | ns             |       |
| sqrt(Si)                           | Disturb          | -0.03 $\pm$ 0.01     | 0.53           | 0.03  |



Table 3. Summary table of concentration vs. discharge plots. High disturbance streams are those with disturbance intensities greater than 7 % of the catchment, low disturbance streams are those with disturbance intensities less than 7 % of the catchment.

| Disturbance Level |            | Direction  | TSS   | SRP   | NH <sub>4</sub> <sup>+</sup> | NO <sub>3</sub> <sup>-</sup> | SO <sub>4</sub> <sup>2-</sup> | DOC  |
|-------------------|------------|------------|-------|-------|------------------------------|------------------------------|-------------------------------|------|
| High              | Count      | Clock.     | 11    | 1     | 4                            | 11                           | 1                             | 1    |
| High              |            | Ant-clock. | 1     | 3     | 3                            | 0                            | 12                            | 3    |
| High              |            | Indeterm.  | 7     | 15    | 12                           | 8                            | 6                             | 2    |
| High              | Proportion | Clock.     | 0.58  | 0.05  | 0.21                         | 0.58                         | 0.05                          | 0.17 |
| High              |            | Anti-clock | 0.05  | 0.16  | 0.16                         | 0.00                         | 0.63                          | 0.50 |
| High              |            | Indeterm.  | 0.37  | 0.79  | 0.63                         | 0.42                         | 0.32                          | 0.33 |
| Total n           |            |            | 19.00 | 19.00 | 19.00                        | 19.00                        | 19.00                         | 6.00 |
| Low               | Count      | Clock.     | 10    | 0     | 4                            | 6                            | 0                             | 1    |
| Low               |            | Anti-clock | 0     | 2     | 3                            | 3                            | 6                             | 6    |
| Low               |            | Indeterm.  | 3     | 11    | 6                            | 4                            | 7                             | 0    |
| Low               | Proportion | Clock.     | 0.77  | 0.00  | 0.31                         | 0.46                         | 0.00                          | 0.14 |
| Low               |            | Anti-clock | 0.00  | 0.15  | 0.23                         | 0.23                         | 0.46                          | 0.86 |
| Low               |            | Indeterm.  | 0.23  | 0.85  | 0.46                         | 0.31                         | 0.54                          | 0.00 |
| Total n           |            |            | 13.00 | 13.00 | 13.00                        | 13.00                        | 13.00                         | 7.00 |



### Figure Captions

Figure 1. Map showing the 10 study catchments located on the Fort Benning Military Installation near Columbus, Georgia. Study catchments include two tributaries of Bonham Creek (BC1, BC2), three tributaries of Sally Branch Creek (SB2, SB3, and SB4); two tributaries of Kings Mill Creek (KM1, KM2); one tributary of Little Pine Knot Creek (LPK); Hollis Branch Creek (HB); and Lois Creek (LC).

Figure 2. Photographs of disturbed upland areas and stream study sites from the Fort Benning Military Installation: A. Heavily disturbed upland area; B. Unpaved road with eroded gullies; C. Stream site from a low disturbance catchment (Lois Creek (LC)); and D. Stream site from a high disturbance catchment (Sally Branch tributary (SB4)).

Figure 3. Seasonal mean discharge and suspended sediment concentrations across all streams. A. Discharge; B. Total suspended sediments (TSS); C. Inorganic suspended solids (ISS); D. Organic suspended sediments (OSS); Separate bars are shown for each year. Error bars are one standard error. Letters indicate where significant differences exist among seasons ( $p < 0.05$ ; Scheffé adjustment for multiple comparisons).

Figure 4. Seasonal mean stream concentrations of A. Dissolved organic carbon (DOC); B. Soluble reactive phosphorus (SRP); C.  $\text{NO}_3^-$ ; D.  $\text{NH}_4^+$ ; E. Specific conductivity; and F. pH. Error bars are one standard error. Letters indicate where significant differences exist among seasons ( $p < 0.05$ ; Scheffé adjustment for multiple comparisons).



Figure 5. Relationship between disturbance and A. Total suspended sediments (TSS); B. Inorganic suspended sediments (ISS) and C. Organic suspended sediments (OSS). Error bars are one standard error. Trend lines are shown for significant relationships ( $p < 0.05$ ). Regressions statistics are shown in Table 2. Stream BC1 was excluded from statistical analyses (see Results).

Figure 6. Plots of dissolved nitrogen concentrations vs. disturbance: A.  $\text{NO}_3^-$ ; B.  $\text{NH}_4^+$ ; and C. Total dissolved inorganic nitrogen (DIN). Error bars are one standard error. Dashed trend line shown for marginally significant relationship ( $p = 0.06$ ); regression statistics are shown in Table 2. Stream BC1 was excluded from statistical analyses (see Results).

Figure 7. Relationship between disturbance and A. Soluble reactive phosphorus (SRP) and B. Dissolved organic carbon (DOC). Error bars are one standard error. Trend lines are shown. Regression statistics are shown in Table 2. Stream BC1 was excluded from statistical analyses (see Results).

Figure 8. Relationships between A. Disturbance and pH; B. Disturbance and  $[\text{Ca}^{2+}]$ ; and C.  $[\text{Ca}^{2+}]$  and pH. Statistics for the pH regression are shown in Table 2. Spearman rank correlations are shown in Panels B and C. Error bars are one standard error. Error bars are not visible for pH (Panel A) because they do not extend beyond data points. Stream BC1 was excluded from statistical analyses (see Results).



Figure 9. Relationship between disturbance intensity and A. [Si]; B. Specific conductivity (cond.); C. [Cl<sup>-</sup>]; and D. [SO<sub>4</sub><sup>2-</sup>]. Error bars are one standard error. Stream BC1 was excluded from statistical analyses (see Results).

Figure 10. Discharge and suspended sediment and solute concentrations in two streams during a storm 29-30 January 2003. Panels A through E are for LC, which drains a relatively low disturbance catchment and panels F through J are for SB3, which drains a relatively high disturbance catchment (see Table 1). The points to the left and separated from the others in panels B-E and G-J represent pre-storm values determined from samples collected about 1 day before the storm. Rainfall totaled 3.3 cm during the period from 20:00 29 September to 16:00 30 September as measured at a site within 10 km of each stream. Dashed line marks peak discharge for the storms. Note that y-axis scale differs between left and right columns for panels A and F.

Figure 11. Example concentration vs. discharge plots for A. total suspended sediments (TSS); B. SO<sub>4</sub><sup>2-</sup> and C. NO<sub>3</sub><sup>1-</sup>. Data are from tributary 2 of Kings Mill Creek (KM2) on 11 July 2002. Arrows point from beginning to end of storm.

Figure 12. Relationship between maximum change in suspended sediment during a storm (=maximum storm concentration – baseflow concentration) and disturbance for A. total suspended solids (TSS); B. Inorganic suspended solids (ISS) and C. Organic suspended solids (OSS); Inset panel shows [OSS] data with finer scale y-axis. Data points are stream averages. Error bars are one standard error. Spearman rank



correlational analysis results are given in each panel. Stream BC1 was excluded from statistical analyses (see Results).

Figure 13. Relationship between maximum change in dissolved nutrient concentrations during a storm (=maximum storm concentration – baseflow concentration) and disturbance for A. soluble reactive phosphorus (SRP) and B.  $\text{NO}_3^-$ . Data shown are stream averages. Spearman rank correlational analysis results are given in each panel. Error bars are one standard error. Stream BC1 was excluded from statistical analyses (see Results).



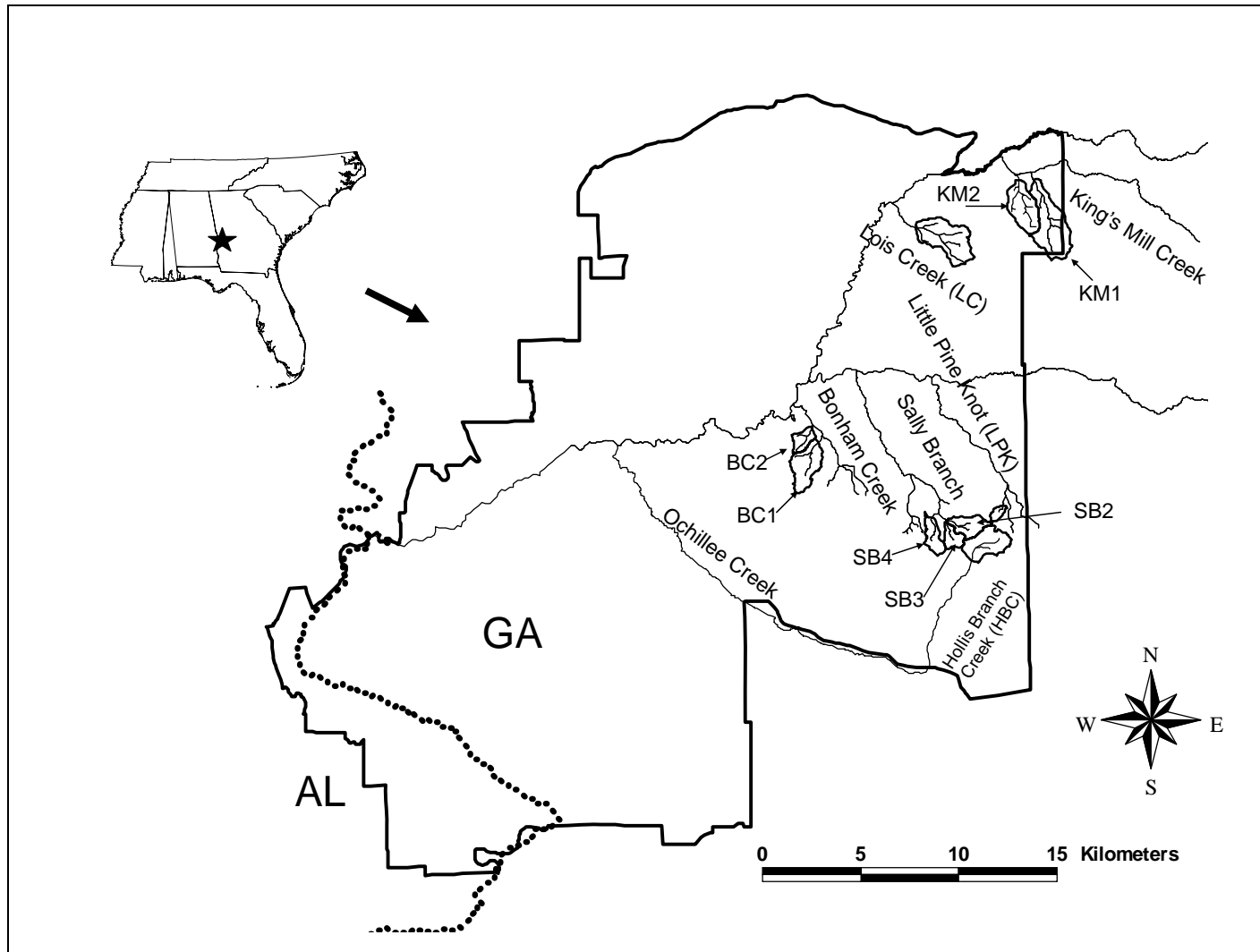


Figure 1



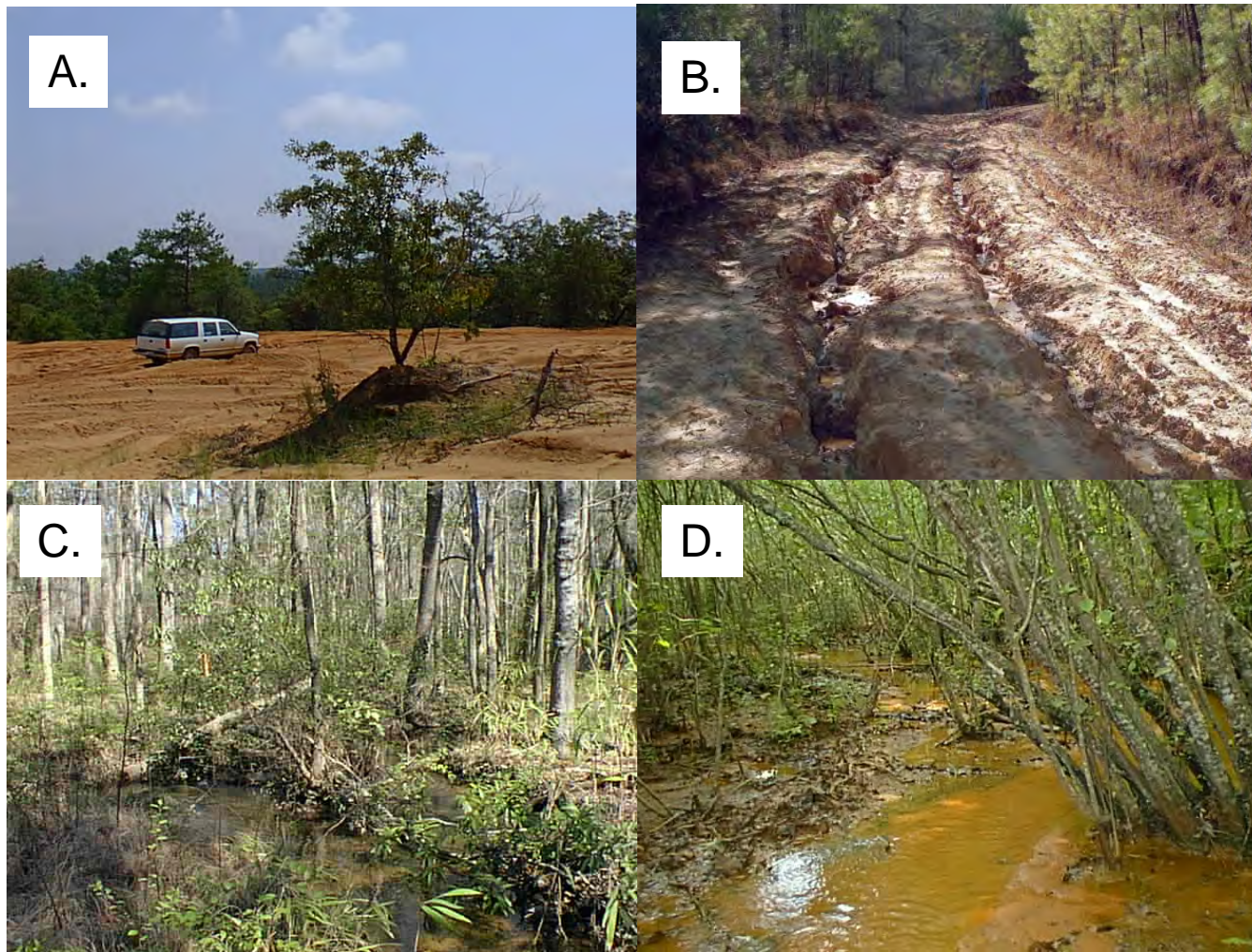


Figure 2



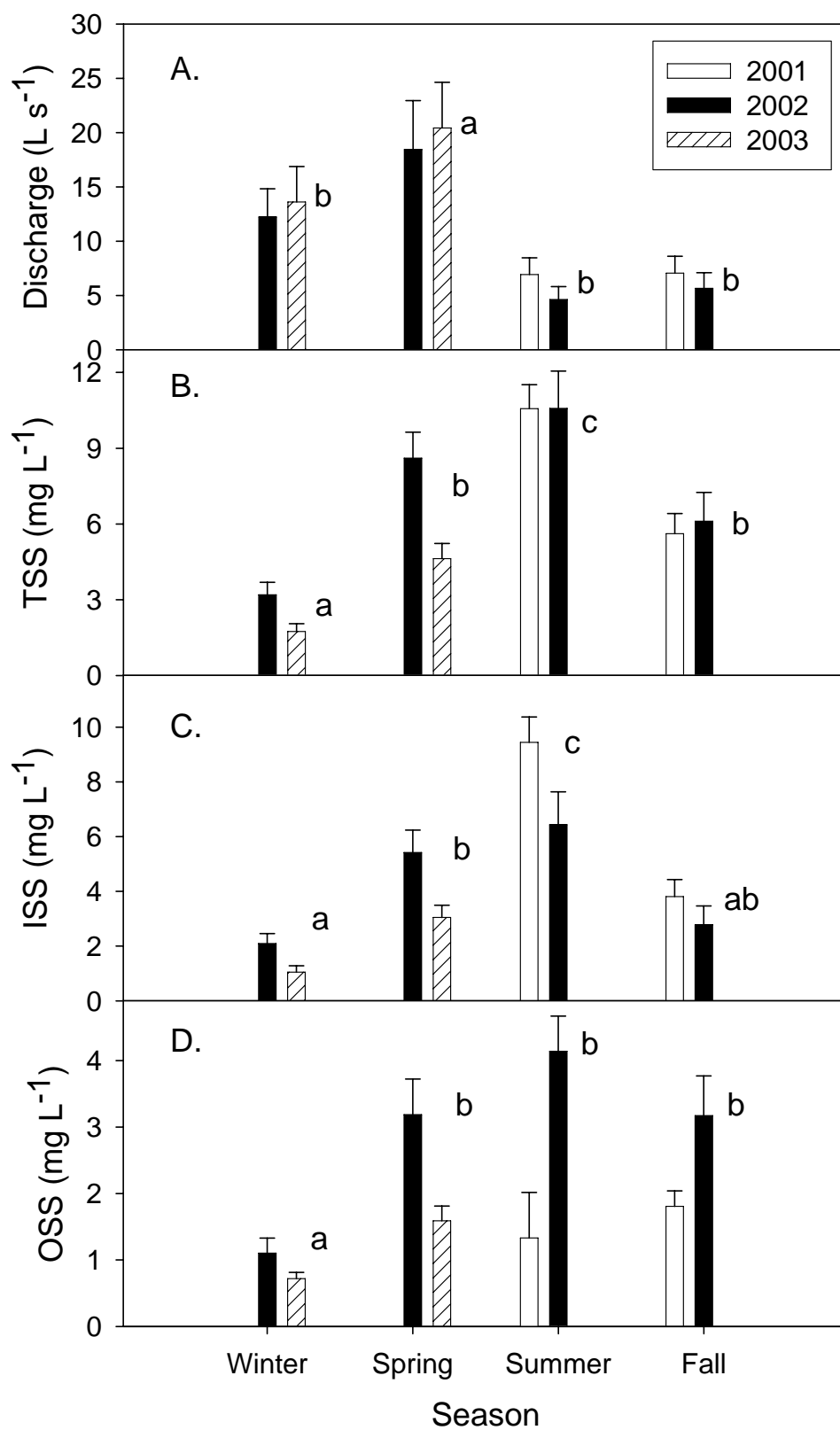


Figure 3



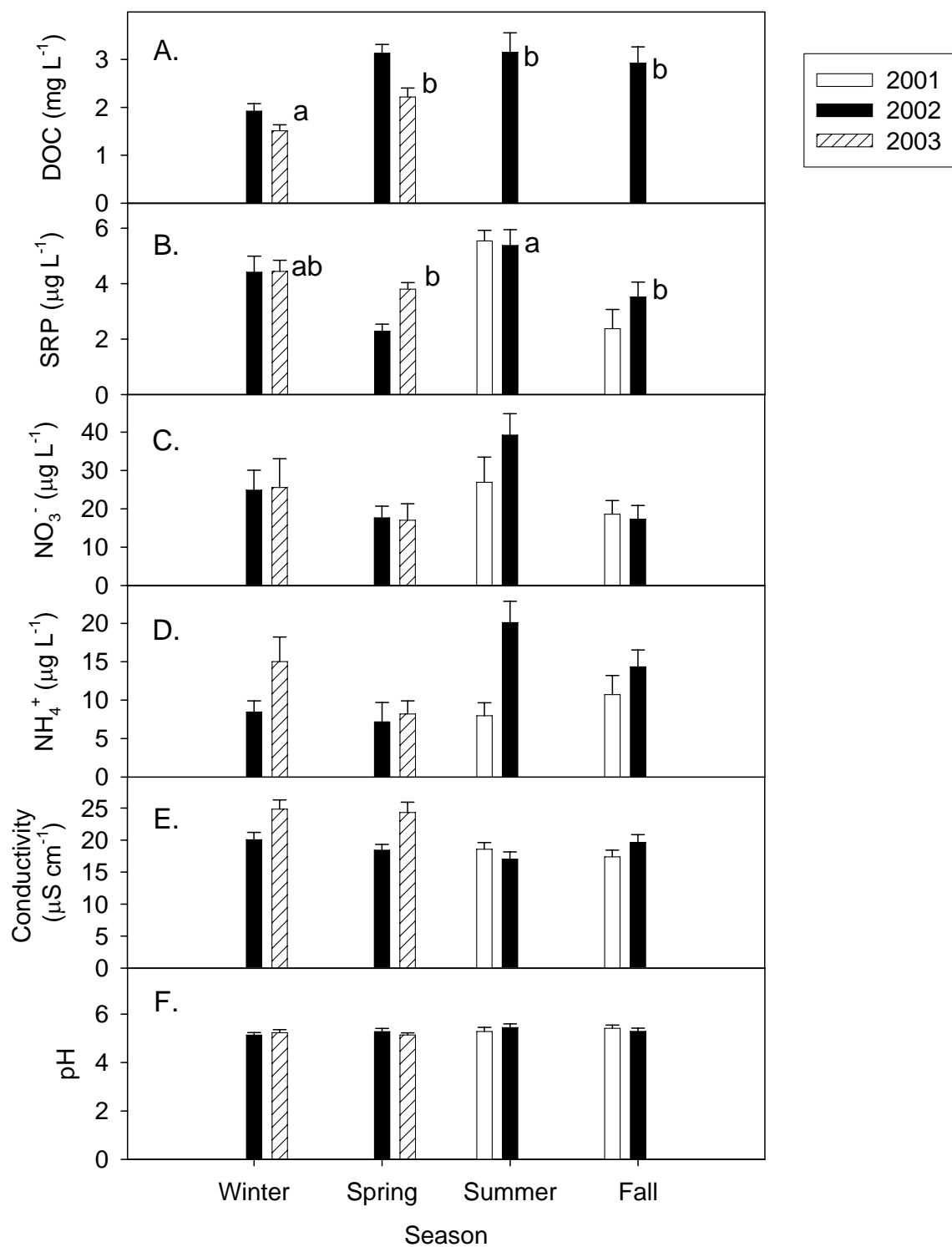


Figure 4



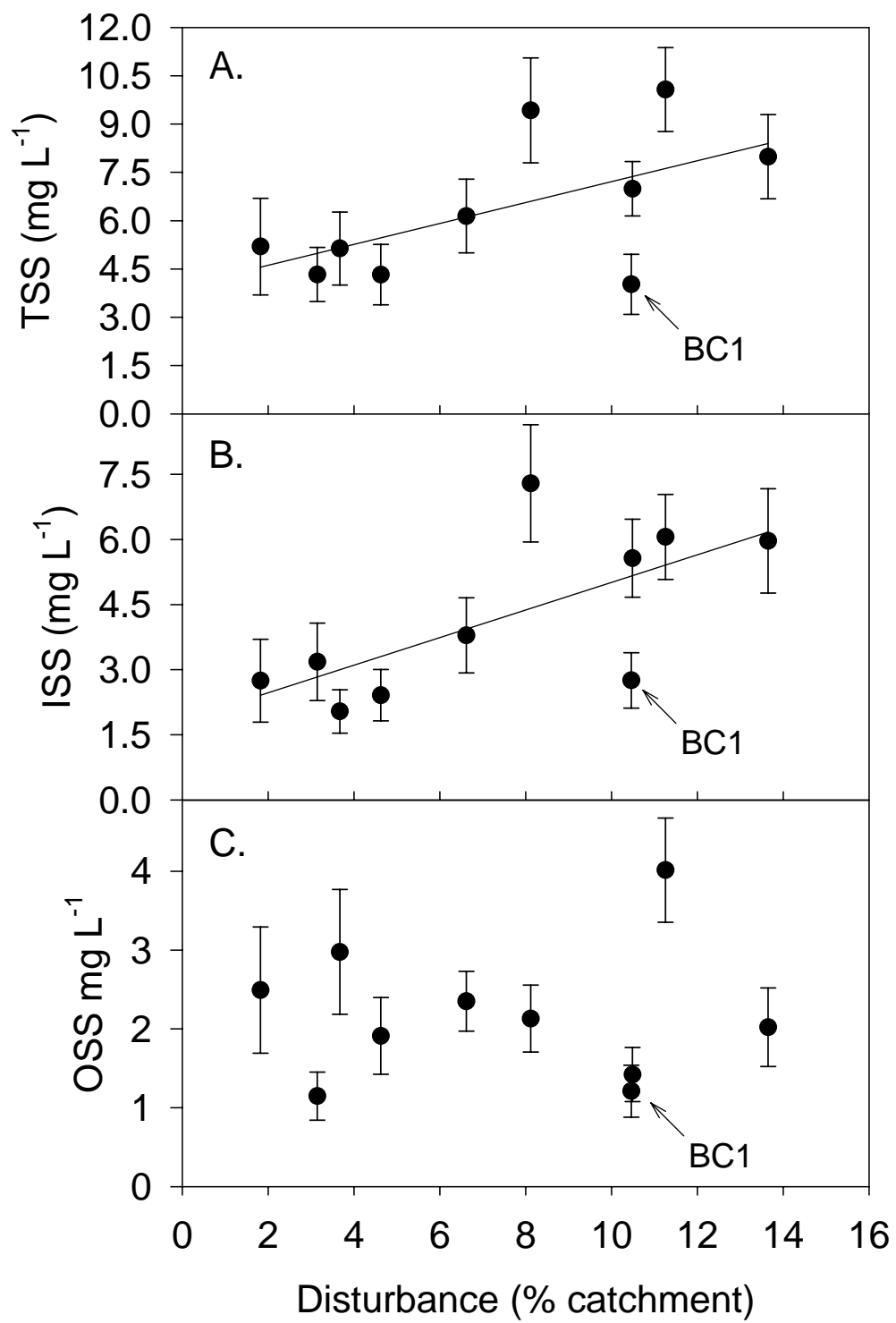


Figure 5



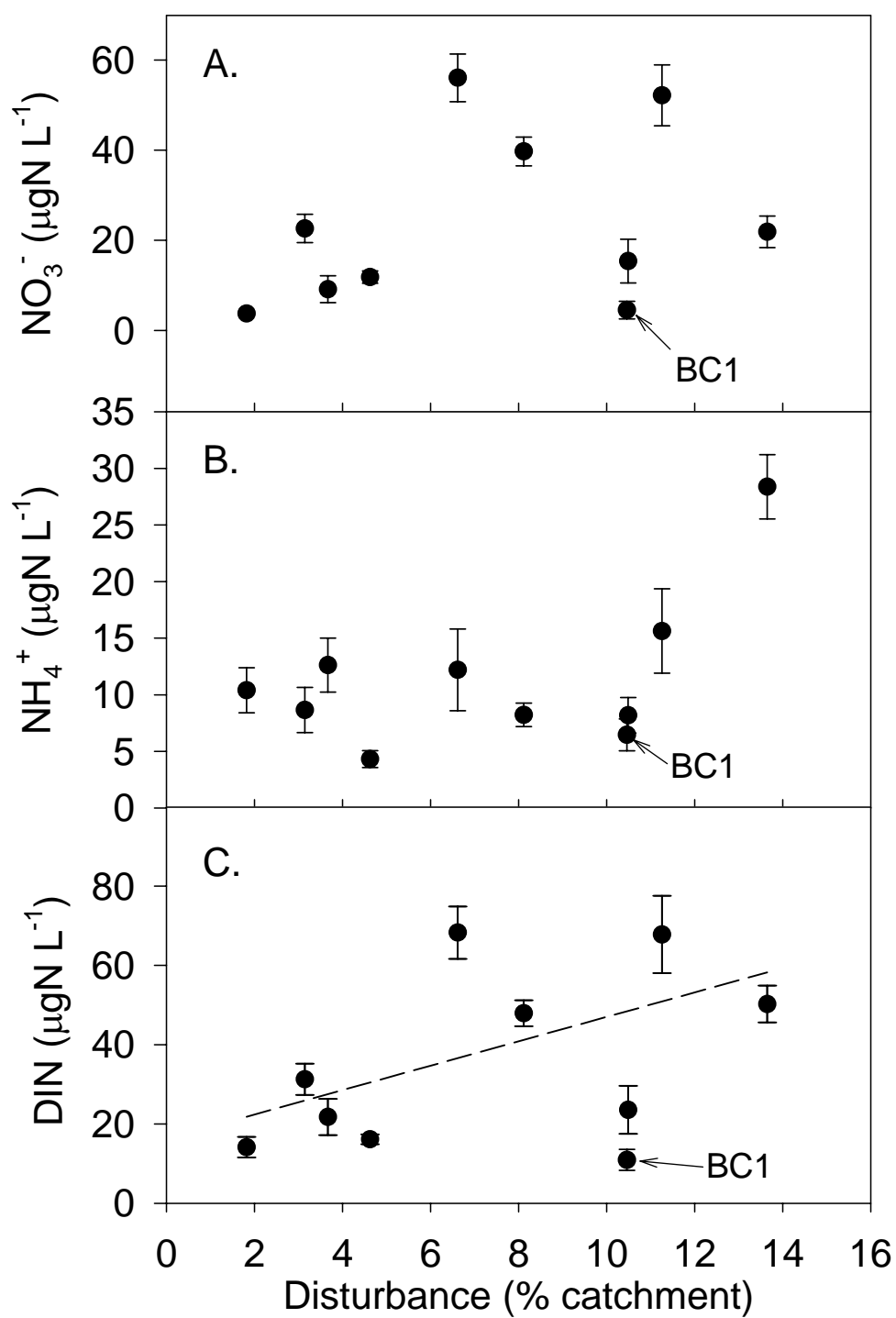


Figure 6



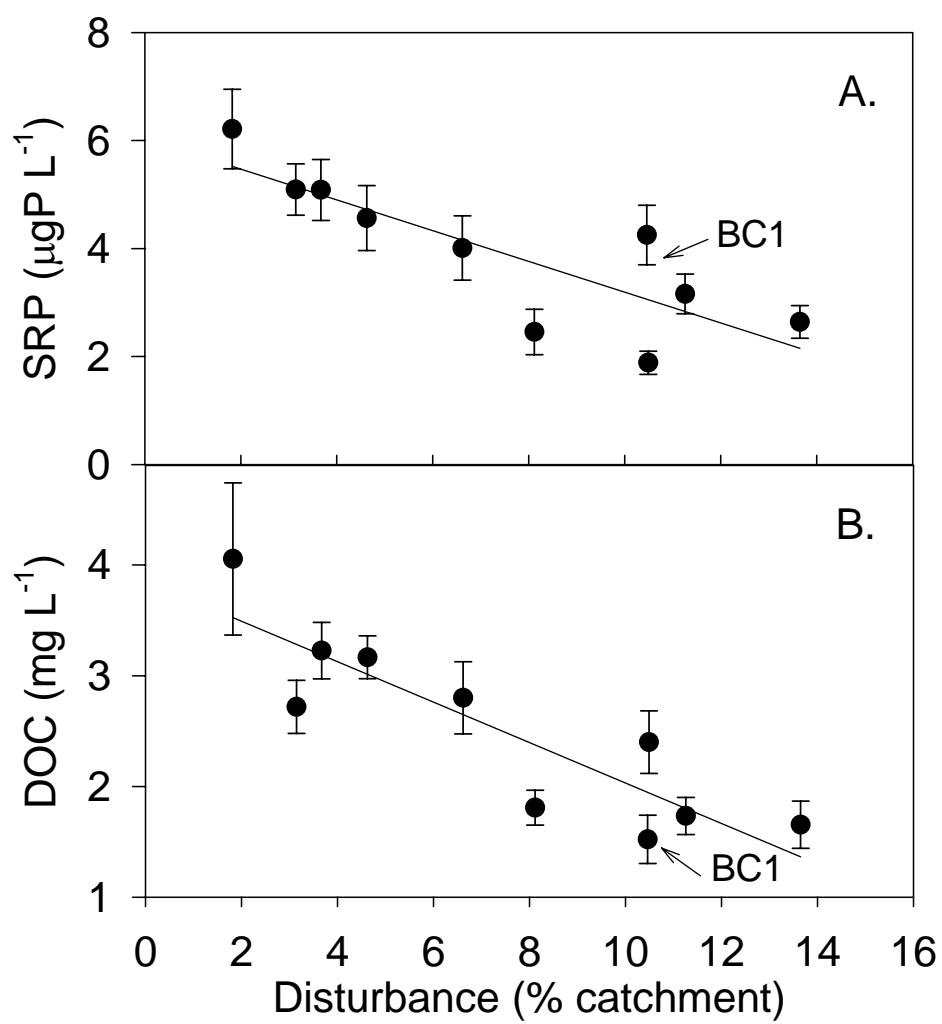


Figure 7



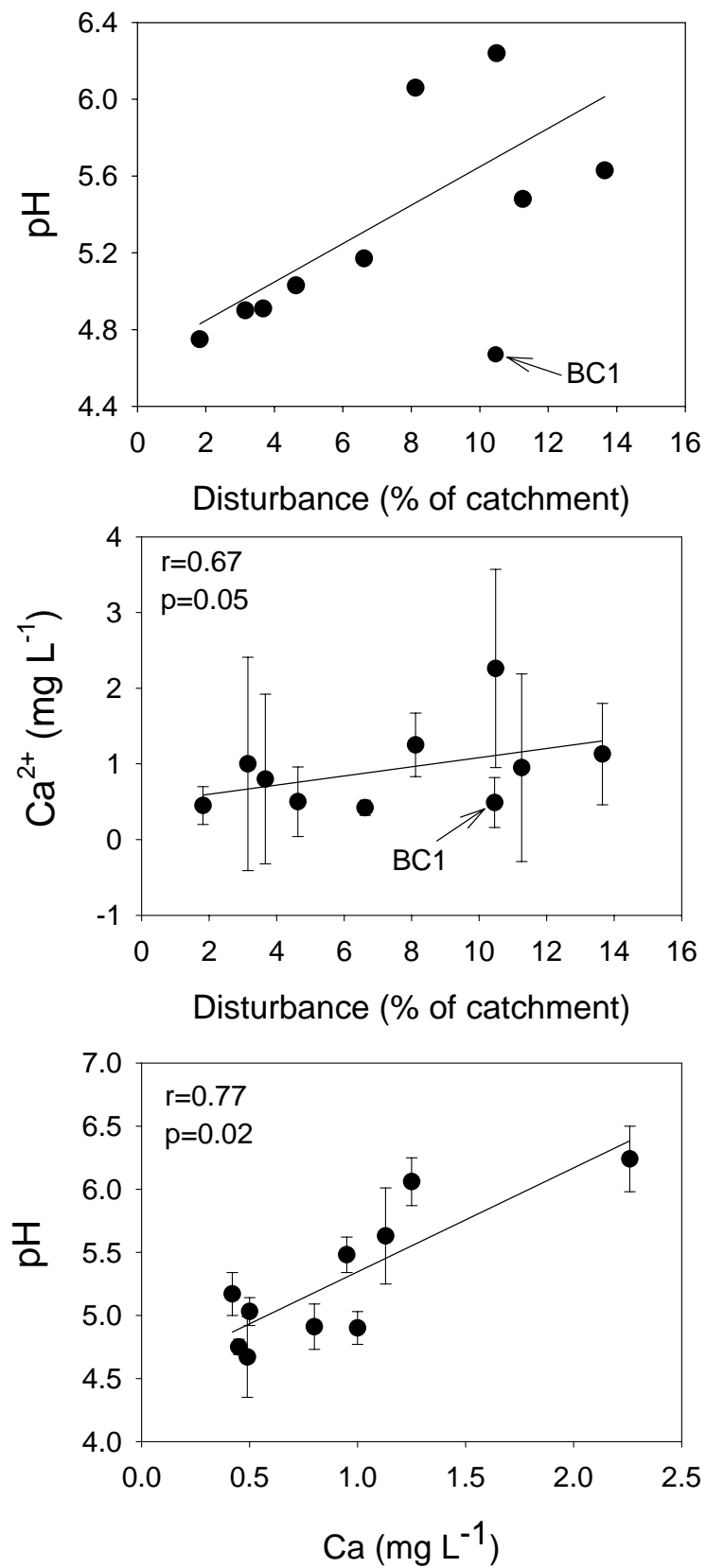


Figure 8



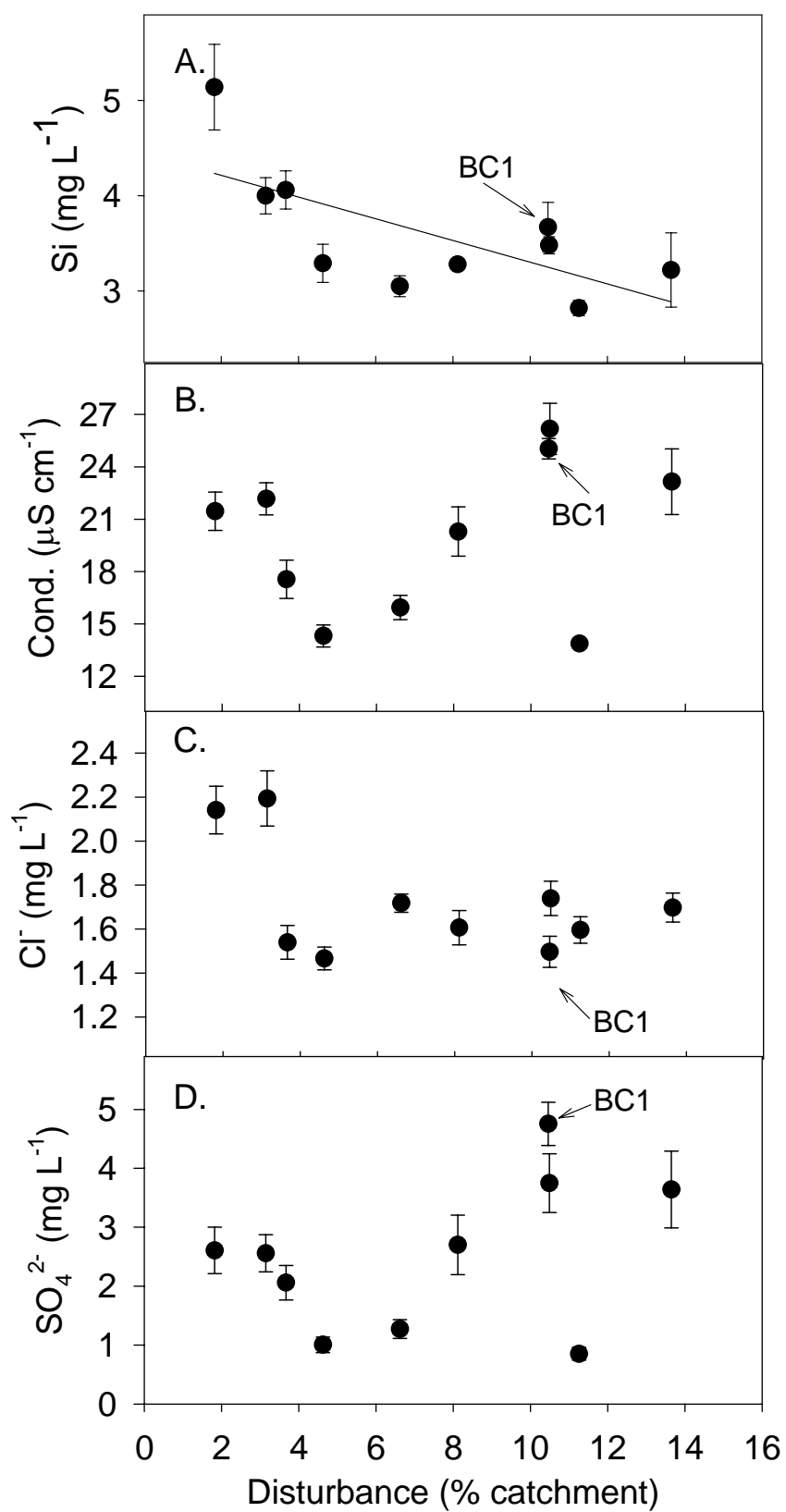


Figure 9



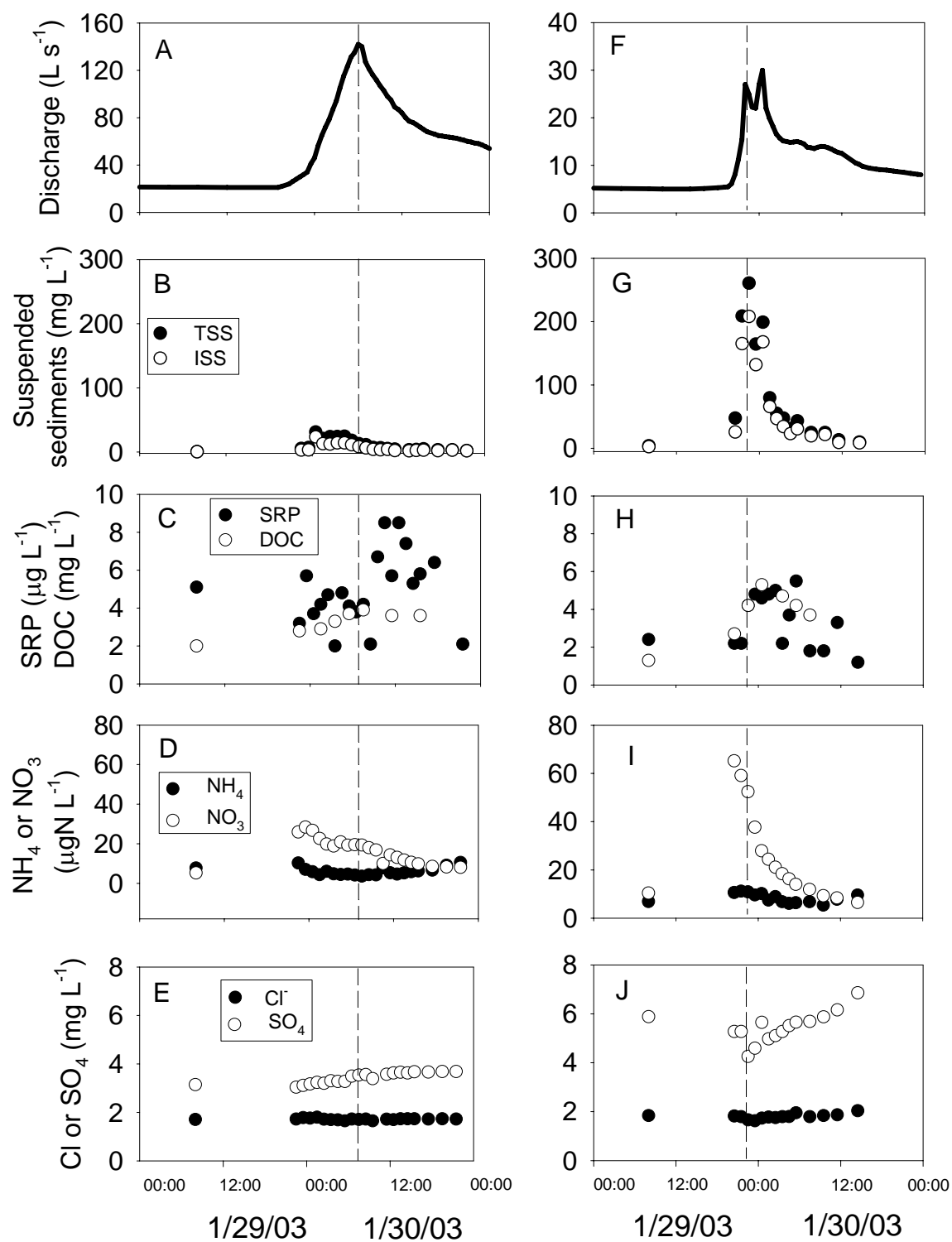


Figure 10



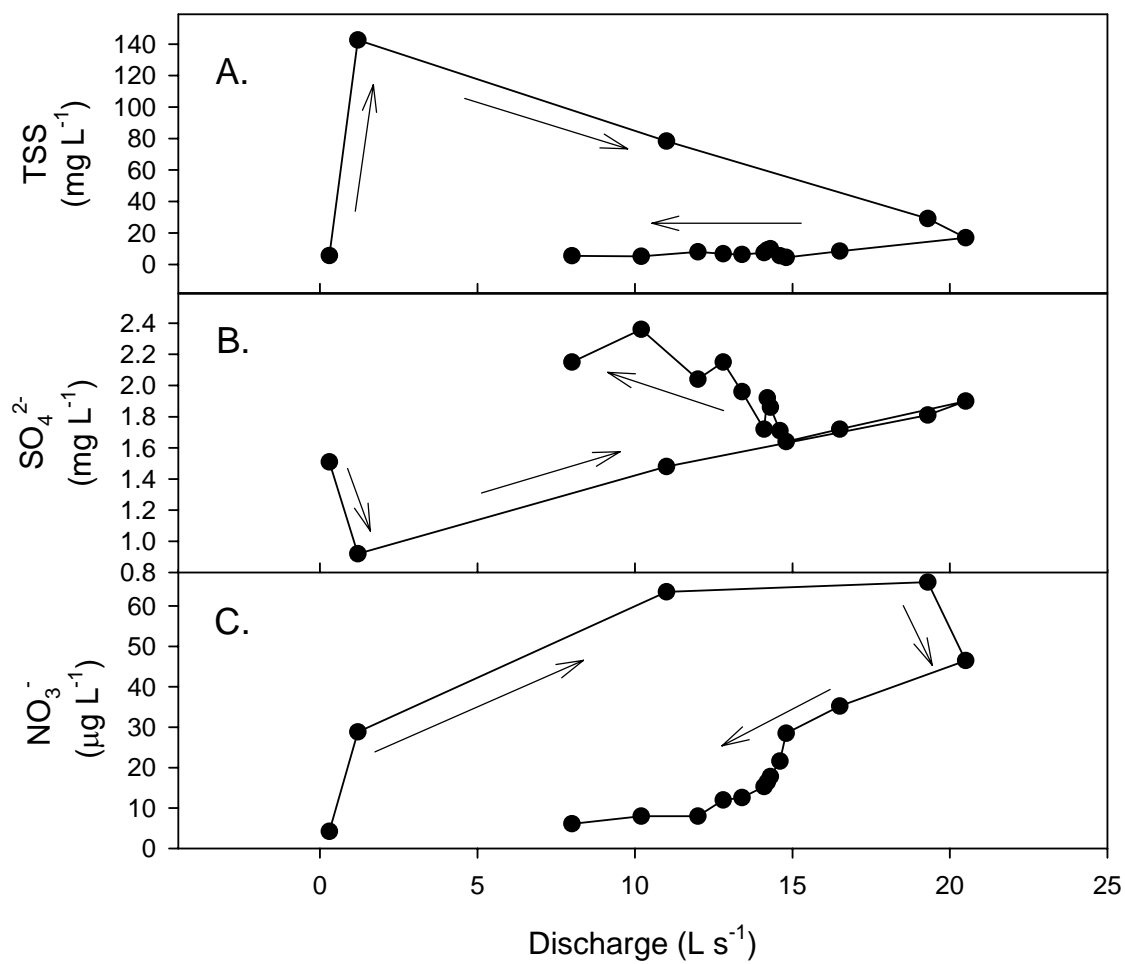


Figure 11



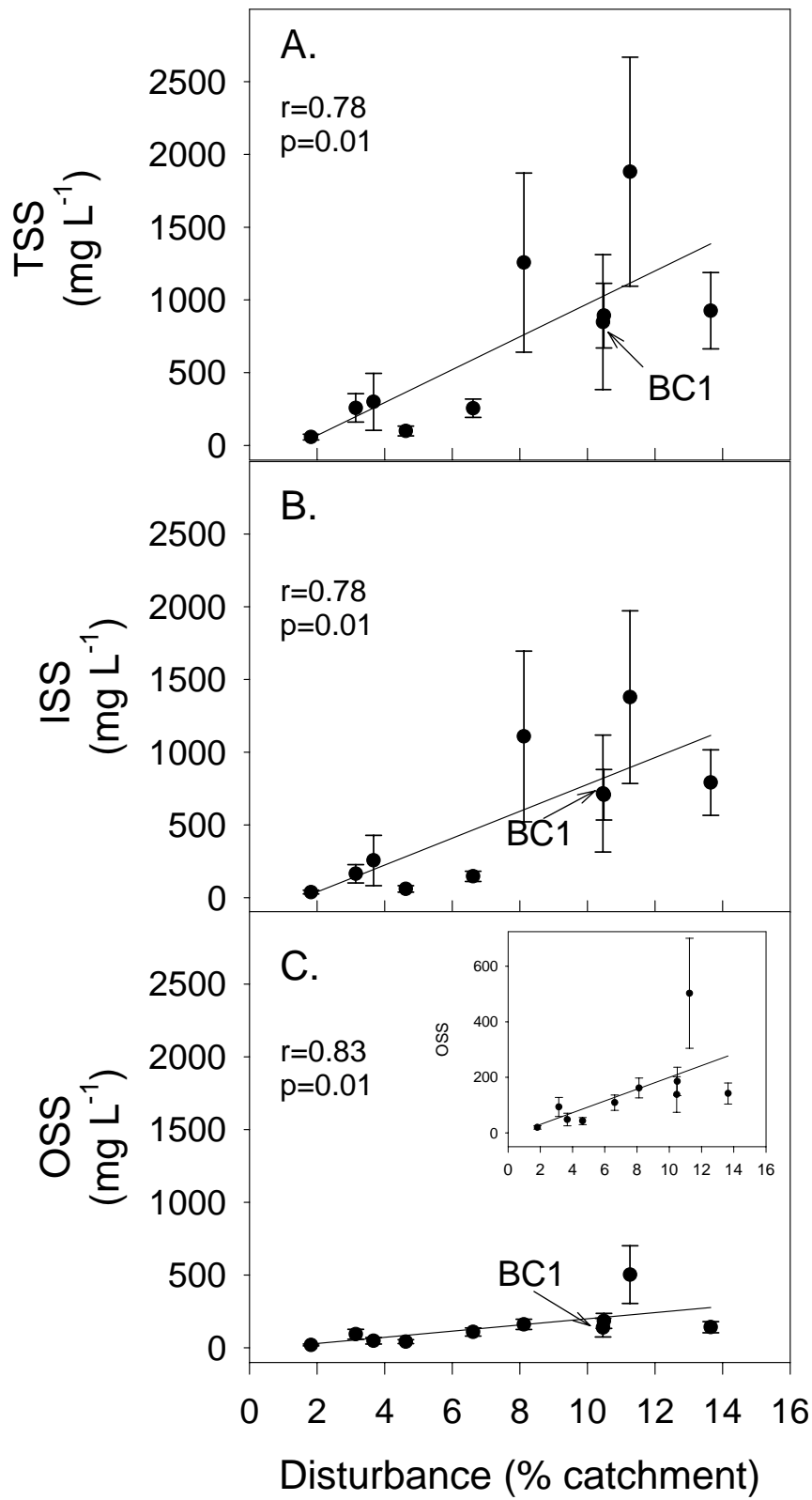


Figure 12



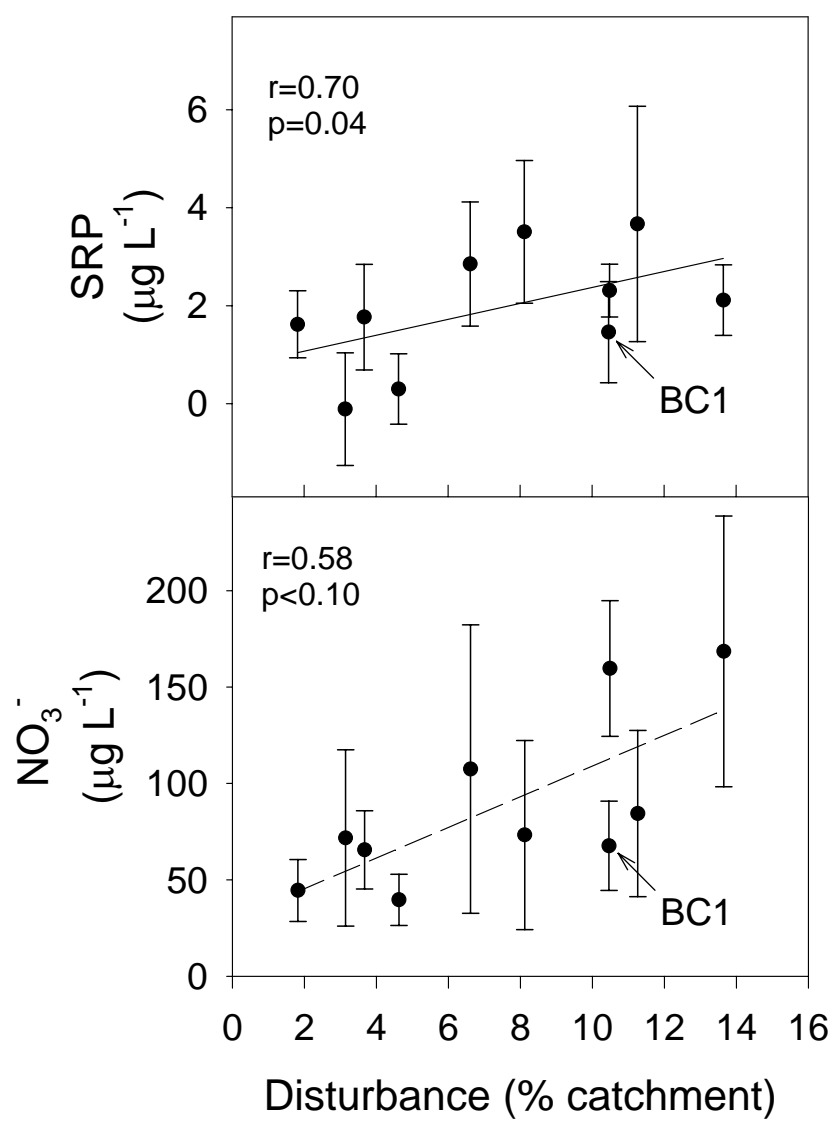


Figure 13



## Appendix C<sup>7</sup> of Indicators Final Report

Stream diurnal dissolved oxygen profiles as indicators of in-stream metabolism and disturbance effects: Fort Benning as a case study

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## Abstract

We investigated whether two characteristics of stream diurnal dissolved oxygen profiles, the daily amplitude and maximum value of the dissolved oxygen saturation deficit, are useful indicators of stream metabolism and the effects of catchment-scale disturbances. The study was conducted at the U.S. Army's Fort Benning installation where vegetation loss and high rates of erosion from intensely-used training areas and unpaved roads have resulted in extensive sedimentation in some streams. Diurnal profiles of dissolved oxygen were measured in ten 2<sup>nd</sup> order streams draining catchments which exhibited a range of disturbance levels. Rates of gross primary production (GPP) and total ecosystem respiration (R) per unit surface area were determined for each stream using the single-station diurnal dissolved oxygen change method with direct measurement of air-water oxygen exchange rates. The daily amplitude of the diurnal dissolved oxygen deficit profile was highly correlated with daily rates of GPP, and multiplying the daily amplitude by average stream depth to account for differences in water volume did not improve the correlation. The daily maximum dissolved oxygen deficit was highly correlated with daily rates of R, and multiplying by average stream depth improved the correlation. In general, these indicators of stream metabolism declined sharply with increasing catchment disturbance level, although the indicators of R showed a more consistent relationship with disturbance level than those of GPP. Our results show that the daily amplitude and maximum value of diurnal dissolved oxygen deficit profiles are good indicators of reach-scale rates of metabolism and the effects of catchment-scale disturbance on these metabolism rates. At Fort Benning, and



presumably at other military installations, they are useful tools for evaluating trends in impacts from military training or rates of recovery following restoration activities.

Key words: stream, diurnal dissolved oxygen, metabolism, indicator, primary productivity, respiration, military



## 1. Introduction

Stream measurements can be good indicators of ecological conditions at the scale of entire watersheds because of the spatially integrating effects of drainage water emerging as streamflow. Various stream chemistry indicators have been used to identify changes in ecosystem function at the watershed scale. For example, stream nitrate concentrations have been used to infer various perturbations of terrestrial nutrient cycling resulting from deforestation (Bormann and Likens 1979), insect defoliation (Swank 1988), N deposition effects (Aber et al. 2003), and climate perturbations (Mitchell et al. 1996, Aber et al. 2002).

Measurements of stream metabolism, if made over entire stream reaches, are potentially good indicators of ecological disturbance at the watershed scale because metabolism is influenced by a combination of physical, chemical, and biological characteristics of streams which, in turn, are affected by drainage water transport from throughout the catchment. However, measurements of reach-scale stream metabolism are laborious, even with automated dissolved oxygen monitors, because they require good estimates or direct measurements of air-water gas exchange and involve many computations (Marzolf et al 1994, McCutchan et al. 1998, Mulholland et al. 2001). Diurnal profiles of dissolved oxygen concentration contain much of the information needed for stream metabolism determinations and may be useful as indicators of stream metabolism.

Stream metabolism indices based on characteristics of diurnal dissolved oxygen profiles have been previously proposed by several investigators. Chapra and Di Toro (1991) developed the “delta method” which uses the average dissolved oxygen deficit,



the daily range in the deficit, and the time of minimum deficit in a graphical approach to estimate respiration rate, gross primary production rate, and reaeration rate, respectively. In a simplification of the delta method, Wang et al. (2003) proposed the “extreme value method” in which the maximum and minimum dissolved oxygen deficits are used to estimate stream metabolism rates. However, both the delta and extreme value methods used in these studies provided volumetric-based estimates of metabolism rates rather than rates per unit of stream surface area, and neither has been evaluated against metabolism rates determined using the diurnal dissolved oxygen approach with direct measurements of the air-water oxygen exchange rate using volatile gas tracers (Marzolf et al. 1994).

Our objectives in this study were to evaluate whether two simple characteristics of diurnal dissolved oxygen profiles, the daily amplitude and maximum value of the dissolved oxygen saturation deficit profile, are useful indicators of: (1) reach-scale stream metabolism rates per unit area, and (2) effects of catchment-scale disturbance to stream ecosystems. We evaluated two versions of each of these potential indicators. The first version was essentially the same as used in the extreme value method of Wang et al. (2003) – the daily amplitude and maximum value of the diurnal dissolved oxygen deficit profile. The second version involved multiplying both the daily amplitude and maximum value by average stream water depth to account for the effect of differences in stream water volume.

The study was conducted at the U.S. Army’s Fort Benning installation where vegetation loss and high rates of erosion from intensively used training areas and unpaved roads have resulted in extensive sedimentation in some streams. In a related paper we show that these catchment-scale disturbances have resulted in reduction in



stream gross primary productivity and total ecosystem respiration rates during some seasons (Houser et al., in review). In this paper we evaluate whether two easily measured characteristics of diurnal dissolved oxygen profiles can be used as surrogate measures or indicators of stream metabolism rates to identify effects of catchment-scale disturbances on important stream functions.

## 2. Study site

Fort Benning is a U.S. Army infantry training base occupying 73,503 ha in central-western Georgia, USA. The climate is humid and mild with rainfall occurring regularly throughout the year and averaging 105 cm annually. Fort Benning is located primarily within the Sand Hills and Southern Hilly Gulf Coastal Plain Level IV ecoregions of the southeastern, U.S. (Griffith et al. 2001). Much of the base is forested and dominated by mixed longleaf (*Pinus palustris*), loblolly (*Pinus taeda*) and shortleaf pine (*Pinus echinata*) communities with some areas of mixed hardwoods dominated by oak (Elliot et al. 1995). Sandy or sandy clay loam soils cover most of the base (Elliot et al. 1995). Streams are relatively low gradient (1 to 2.5% slope) and often are meandering within broad floodplains. Floodplain vegetation is primarily mesic hardwoods dominated by water oak (*Quercu nigra*), sweetgum (*Liquidamber styraciflua*), and swamp tupelo (*Nyssa sylvatica* var. *biflora*) (Cavalcanti 2004). Streams have generally sandy bottoms but often contain roots of riparian vegetation and considerable amounts of woody debris and deposits of fine organic matter, particularly if catchments are relatively undisturbed.



Fort Benning contains numerous areas ranging from several to hundreds of ha in size used intensively for military training activities involving tracked and other vehicle maneuvering (Dale 2002). These areas, which are primarily in the upland portions of catchments, have become denuded of most vegetation and the soils are highly disturbed and subject to extensive erosion. Ephemeral drainages convey the eroded sediments to perennial streams where subsequent sedimentation has resulted in burial of organic matter and highly unstable, shifting bottom sediments low in organic matter content. In addition unpaved roads also experience high rates of erosion and contribute large quantities of eroded sediments to streams.

### **3. Methods**

Ten 2<sup>nd</sup> order streams draining catchments with a range of disturbance levels were chosen for study (Figure 1). Total catchment area and disturbance level in each catchment were quantified using digital elevation maps for Fort Benning (<http://sempdata.wes.army.mil>) and GIS-based land cover maps developed from satellite imagery (Dale et al. 2004). Catchment disturbance level was defined as the sum of the proportion of the catchment consisting of bare ground on slopes > 3% and the proportion of the catchment covered by unpaved roads (see Maloney et al. in review for additional details). Except for four highly disturbed catchments, disturbance levels for all 249 2<sup>nd</sup>-order stream catchments at Ft. Benning span a range of 0 to 17 %. The 10 catchments included in this study spanned much of this available disturbance gradient with disturbance levels ranging from 1.8 – 13.6%.



Reach-scale rates of gross primary production (GPP) and total ecosystem respiration (R) were determined for each stream using the open-channel, single-station diurnal dissolved oxygen change method (Owens 1974, Bott 1996). YSI model 6000 or 600 series sondes equipped with YSI model 6562 dissolved oxygen probes were deployed for 2 to 3 week periods in a well mixed point in each stream during the period from 8 April to 12 May 2002; recording dissolved oxygen concentrations and water temperatures at 15-min intervals. The sondes were deployed in only five streams at any one time due to equipment limitations. To reduce variability resulting from weather-related effects, data only for those dates with relatively high values of daily photosynthetically-active radiation ( $>70\%$  of the seasonal maximum value as measured at a meteorological station within 10 km of the streams) and discharge values similar to those during sonde deployment were used in the analysis.

At the time of sonde deployment, discharge and air-water gas exchange rate were measured in each stream by a simultaneous, steady-state injection of a conservative (NaCl) and volatile (propane) tracer to stream water (Marzolf et al. 1994). Specific conductance and dissolved propane concentrations were measured at two stations. The upper station was about 10-20 m downstream from the injection to ensure complete mixing of the injected solution and gas. The downstream station was located 40 to 100 m downstream from the upper station, depending on the water velocity. Specific conductance was measured using a YSI model 30 conductivity meter. Stream discharge was calculated from the difference between pre-injection and steady state values of specific conductance in the stream, the specific conductance of the injection solution, and the injection rate. Average water travel time for each study reach was determined from



the time of maximum slope of the rate of increase in specific conductance at each station. Average water velocity was calculated from the water travel time and the distance between sampling stations. Average wetted width of the stream was determined by measuring widths every 5 m between the two sampling stations. Average water depth was computed as discharge divided by the product of average wetted width and average water velocity.

For dissolved propane measurements, 6 samples were collected at each station by drawing 6 mL of stream water into a syringe and injecting the water into pre-evacuated 8 mL vials. The headspace in the vials was then equilibrated with the atmosphere and the vials returned to the laboratory for analysis. Propane concentrations in subsamples of headspace gas (80  $\mu$ L) were determined within 1-2 days of collection by gas chromatography using a Hewlett Packard Model 5890 Series II with a Poraplot Q column and flame ionization detector. Air-water exchange rate of propane was calculated as the first-order rate constant of propane concentration decline from the upper to the lower station, corrected for dilution due to increase in discharge, and divided by water travel time between the two stations. Propane gas exchange rates were then converted to oxygen gas exchanges rates by multiplying by a factor of 1.39 (Rathbun et al. 1978).

Streamwater pH, dissolved organic carbon (DOC) and nutrient concentrations were also determined at the time of sonde deployment in each stream. Methods for these analyses are presented in Maloney et al. (in press).

Rates of net dissolved oxygen change due to metabolism (net metabolism) were determined as the difference between successive 15-min measurements corrected for air-water oxygen exchange between measurements and average water depth. The rate of air-



water oxygen exchange was calculated as the product of the rate coefficient (determined from the NaCl/propane injections) and the observed % dissolved oxygen saturation using barometric pressure data from a meteorological station within 10 km of the stream and converted to stream elevation. Nighttime R was calculated as the sum of the net metabolism measurements during the night. Daytime R was determined by interpolating between respiration rates measured 1 hour before dawn and 1 hour after dusk. Total daily R was the sum of nighttime and daytime R over 24 h periods (from midnight to midnight). Daily GPP was the sum of the differences between the interpolated daytime respiration rates and the observed net metabolism (see Figure 2 and Marzolf et al. 1994 for a more complete explanation of the calculations of daily GPP and R).

We calculated the daily amplitude and maximum absolute value for each of the diurnal dissolved oxygen saturation deficit profiles as indices of daily GPP and R, respectively. We used 1-hour running average values of the dissolved oxygen saturation deficit for these calculations to avoid bias by isolated high or low measurements. We also multiplied the daily amplitude and maximum values by average water depth to provide a second version of each of these indices that takes into account differences in water volume. Average values of these four indices were calculated for each stream over the study period and compared to average values of daily GPP and R as well as to catchment disturbance levels to determine their usefulness as indicators of stream metabolism and response to disturbance.

#### **4. Results**



Stream and catchment characteristics, including disturbance levels, are given in Table 1. Catchments ranged in size from 33 to 369 ha and varied in discharge from 6.9 to 55 L/s. Streams ranged in pH from 4.8 to 6.1 and ionic strength, dissolved organic carbon and nutrient concentrations were relatively low in all streams.

The dissolved oxygen profile for site BC2 on 18 April 2002 shows the typical diurnal pattern observed in these streams (Figure 2A). The diurnal pattern in the dissolved oxygen saturation deficit showing the amplitude and maximum deficit is shown in Figure 2B. Rates of GPP and R are calculated from the areas under the daytime net metabolism peak and the between zero and the net metabolism curve, respectively (Figure 2C).

Correlations between both versions of the amplitude-based indicator and rates of GPP per unit surface area were high (Figure 3A and B). Similarly, correlations between both versions of the maximum deficit-based indicator and R per unit surface area were also high (Figure 3C and D). It appears that accounting for water depth provides a slightly improved indicator for R, but not for GPP.

In general, all of the indicators of metabolism declined with increasing catchment disturbance level (Figure 4). However, the relationship between the daily amplitude and disturbance level was very weak, with disturbance level explaining only 26% of the variation in this version of this indicator (Figure 4A). Relationships between the daily amplitude times water depth and disturbance level and between both versions of the daily maximum deficit indicator and disturbance level were considerably stronger (Figures 4B, C, and D). There was one stream (BC1), however, that appeared to be an outlier in the overall pattern, with considerably higher values for the metabolism indicators than



expected for its catchment disturbance level. Although the catchment of this stream has a relatively high disturbance level, the stream is bordered by a broad forested floodplain and the channel morphology appears to be relatively undisturbed. Excluding BC1 from the analysis, exponential models based on catchment disturbance level explained 76% to 80% of the variation in the amplitude times water depth and both versions of the maximum deficit indicator. With the exception of BC1, average values of indicators adjusted for water depth were very low for streams in catchments with disturbance levels of 8% or more (Figures 4B and D).

## **5. Discussion**

Our results show that diurnal profiles of the dissolved oxygen saturation deficit reflect the rates of in-stream metabolism (GPP and R) and are useful indicators for comparisons of metabolism among streams. The maximum absolute value of the dissolved oxygen deficit (usually occurring early in the nighttime period) is influenced primarily by R and air-water exchange of oxygen, whereas the minimum value of the dissolved oxygen saturation deficit (usually occurring in late morning or early afternoon) represents a balance between GPP, R and air-water exchange. Thus, the daily amplitude of the diurnal deficit profile provides an index of daily GPP and the maximum absolute value of the deficit profile is an index of daily R (Figure 2).

Most of the metabolism within small streams and rivers is associated with the bottom or stationary substrata (e.g., woody debris); however, determination of metabolism rates is usually based on measurements of dissolved oxygen concentration changes in the overlying water. Therefore, we had anticipated that accounting for



differences in water depth would improve correlations between the two oxygen profile-based indicators and rates of metabolism per unit stream area. However, multiplying by average water depth resulted in only a small improvement in the correlation between the maximum deficit and R and a slightly poorer correlation between daily amplitude and GPP (Figure 3).

The apparent lack of effect of differences in water depth on the metabolism indicators may be because air-water exchange of oxygen is also a surface process and tends to counteract the effect of metabolism on streamwater dissolved oxygen concentrations. The primary factors regulating air-water gas exchange rates are surface turbulence and volumetric water mixing. Differences in surface turbulence among our streams were likely small because stream channel gradients were low and not greatly different among the streams (Table 1). Rates of air-water oxygen exchange (expressed as volumetric turnover rates in units of  $\text{time}^{-1}$ ) were negatively correlated with average stream water depths ( $r = -0.82$ ,  $P = 0.004$ , Table 1). Thus, although a given rate of metabolism would be expected to result in smaller changes in streamwater dissolved oxygen concentration in a deeper stream, the volumetric air-water exchange rate of oxygen would also be lower in a deeper stream and counteract, to some degree, the effect of water depth on metabolism.

Our results also show that metrics based on diurnal profiles of the dissolved oxygen saturation deficit in streams are useful indicators of catchment-scale disturbance. Although the daily amplitude of the diurnal oxygen deficit was poorly related to catchment disturbance level, when multiplied by average water depth the relationship between this metric and disturbance level was relatively strong (Figure 4B). The



relationship between the daily maximum deficit value and disturbance level was quite strong with and without the inclusion of average water depth in the calculation of this indicator (Figure 4C and D).

The reason for the contrast in results for the two versions of the disturbance indicator based on the daily amplitude is not clear, particularly when considering that both versions of this indicator were shown to be relatively good indicators of stream GPP. This apparent contradiction may be a result of the relatively low values of GPP in these streams. In all but one stream, GPP was  $< 0.5 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$  which falls in the lower part of the range of values reported for other small streams in the intersite study of Mulholland et al. (2001). Houser et al. (in press) reported a significant negative relationship between GPP and catchment disturbance level only during spring of 2002 during two years of quarterly measurements in these Fort Benning streams. Values of R were generally an order of magnitude higher than GPP and Houser et al. (in press) reported significant negative relationships between R and catchment disturbance level during all seasons but autumn. Thus, diurnal oxygen profile metrics related to R (i.e., daily maximum deficit with and without inclusion of water depth) appear to be much stronger indicators of the effect of catchment disturbance at Fort Benning than metrics related to GPP.

The stream diurnal dissolved oxygen profile metrics are good indicators of catchment-scale disturbance at Fort Benning because the major disturbances to streams at Fort Benning are primarily a result of the effects of increased runoff and erosion from denuded lands in upland portions of catchments rather than direct modification of stream conditions. The effect of upland erosion on streams is produced by sediment transport



during storms and subsequent deposition in stream channels resulting in highly unstable streambeds of shifting sand. Maloney et al. (in press) reported that the amount of coarse woody debris exposed on the stream bottom and the organic matter content of streambed sediments are considerably lower in streams in more highly disturbed catchments at Fort Benning. In addition, the stream channels in many of the highly disturbed catchments appeared to be highly incised, likely the result of channel erosion and downcutting from higher stormflows due to reduced infiltration rates and greater surface runoff. Maloney et al. (in press) showed that stream storm hydrographs are steeper in more disturbed Fort Benning catchments. These effects on the physical characteristics of stream channels in more highly disturbed catchments would tend to reduce the available habitat for colonization and growth of algae and heterotrophic microbes which require stable substrata and organic matter, respectively. Finally, concentrations of soluble reactive phosphorus and DOC in stream water declined significantly with increasing disturbance levels in these catchments, presumably because of increased sorption by clay-rich stream sediments or reduced leaching losses from upland areas denuded of vegetation (Maloney et al. in press). These changes in the physical and chemical characteristics of streams resulting from upland disturbance would be expected to reduce rates of stream metabolism and our indicators of stream metabolism show this.

One stream, BC1, appeared to be an outlier in the pattern of reduced metabolism with increasing catchment disturbance (Figure 4). Although the catchment of this stream has a relatively high disturbance level, the stream channel appears to be much less affected physically by the catchment disturbance. This stream has a very broad, forested floodplain and the abundance of coarse woody debris and the organic matter content of



its sediments are more typical of the less disturbed catchments (Maloney et al., in press). The rates of metabolism (particularly R) in this stream are also more typical of streams in less disturbed catchments (Houser et al., in press).

This outlier suggests caution when using stream metabolism metrics as indicators of catchment-scale disturbance even within small regions. Local factors related to riparian vegetation (e.g., stature and density, leaf phenology, quantity and quality of organic inputs) and floodplain/channel hydraulics can have large effects on stream metabolism. In a study investigating land use effects on stream metabolism, Young and Huryn (1999) showed that GPP was most influenced by characteristics that influenced the availability of light (e.g., shading by the riparian forest canopy, turbidity from disturbed catchments) and R was influenced primarily by characteristics controlling organic matter supply (e.g., riparian leaf input, organic content of sediments). Mulholland et al. (2001), in a study of stream metabolism across different biomes, found that light availability as regulated by riparian vegetation was the primary factor controlling GPP and the extent of surface/subsurface water exchange within stream channels was an important factor controlling R. These riparian and channel characteristics may over-ride or partially obscure effects of catchment-scale disturbances and must be considered when using stream metabolism indicators to assess disturbance impacts.

Our approach using the daily amplitude and maximum value of the diurnal dissolved oxygen deficit profile as indicators of stream metabolism supports the findings of Wang et al. (2003) and earlier those of Chapra and Di Toro (1991). Wang et al. (2003) termed their approach the extreme value method and used it to show impacts of land use change on stream metabolism. They reported that rates of GPP and R derived from these



indicators were higher in an agricultural catchment stream than a stream in an urbanized catchment. Our results expand on these earlier studies by documenting the strong relationship between these indicators and simultaneous measurements of reach-scale rates of GPP and R per unit surface area in streams. Our results further show that these diurnal oxygen profile metrics can be good indicators of catchment-scale disturbance because stream metabolism is influenced by physical, chemical, and biological characteristics of streams which are, in turn, affected by drainage water transport from all portions of the catchment. With the availability of relatively low-cost, data-logging, field dissolved oxygen monitors, these indicators are considerably easier to measure than rates of stream metabolism which require good estimates of air-water gas exchange and involve many computations. At Fort Benning and other military installations, they provide a useful tool for evaluating trends in impacts from military training or rates of recovery following restoration activities.



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Table 1. Stream and catchment characteristics. Data are from measurements made at the beginning of the period of dissolved oxygen measurements. Catchment area, disturbance level and channel gradient are from Maloney et al. (in press).

| Stream                       | Abbreviation | Catchment area (ha) | Disturbance level (%) | Channel Gradient (%) | Discharge (L/s) | Ave Water Depth (cm) | Rearation Rate (min <sup>-1</sup> ) | Specific conductance (μS/cm) | pH   | DOC (mg/L) | DIN (μgN/L) | SRP (μgP/L) |
|------------------------------|--------------|---------------------|-----------------------|----------------------|-----------------|----------------------|-------------------------------------|------------------------------|------|------------|-------------|-------------|
| Kings Mill Creek Tributary 2 | KM2          | 231                 | 1.8                   | 1.00                 | 22.2            | 19.1                 | <0.01                               | 24.4                         | 4.8  | 2.5        | 2.7         | 2.2         |
| Bonham Creek Tributary 2     | BC2          | 74.9                | 3.2                   | 2.67                 | 9.9             | 12.9                 | .020                                | 22.8                         | 4.9  | 3.4        | 16.6        | 3.1         |
| Lois Creek                   | LC           | 332                 | 3.7                   | 1.69                 | 19.1            | 17.1                 | .020                                | 19                           | 4.87 | 2.8        | 0.7         | 2.2         |
| Kings Mill Creek Tributary 1 | KM1          | 369                 | 4.6                   | 0.83                 | 55              | 20.8                 | .026                                | 16                           | 4.97 | 3.7        | 27.2        | 1.8         |
| Hollis Branch                | HB           | 215                 | 6.6                   | 2.00                 | 23.3            | 13                   | .025                                | 20.5                         | 5.05 | 5.4        | 27.2        | 0.3         |
| Sally Branch Tributary 2     | SB2          | 123                 | 8.1                   | 2.31                 | 15.2            | 4.2                  | .044                                | 21.7                         | 6.03 | 2.7        | 36.7        | 0.3         |
| Bonham Creek Tributary 1     | BC1          | 210                 | 10.5                  | 1.67                 | 18.5            | 18.1                 | .012                                | 26.1                         | 4.55 | 3          | 7.2         | 2.4         |
| Sally Branch Tributary 3     | SB3          | 71.7                | 10.5                  | 1.00                 | 6.9             | 6.6                  | .072                                | 18                           | 6.08 | 3.5        | 5.3         | 2.4         |
| Little Pine Knot Tributary   | LPK          | 33.1                | 11.3                  | 5.10                 | 7.2             | 7.1                  | .087                                | 13.4                         | 5.22 | 2.7        | 31.7        | 1           |
| Sally Branch Tributary 4     | SB4          | 100                 | 13.6                  | 1.33                 | 7.4             | 3.5                  | .071                                | 18.4                         | 5.41 | 2.6        | 29.8        | 2.8         |



## List of Figures

Figure 1. Map of Fort Benning showing location of the study catchments.

Figure 2. Diurnal profiles (1-hour running averages) of (A) dissolved oxygen concentration (DO), (B) dissolved oxygen saturation deficit showing the amplitude and maximum absolute value of the deficit profile, and (C) net metabolism rate showing the area representing gross primary production (GPP, vertical lines) and total ecosystem respiration (R, horizontal lines). Data are from 18 April 2002.

Figure 3. Relationships between gross primary production rate (GPP) and (A) amplitude of the diurnal dissolved oxygen (DO) deficit profile, and (B) amplitude of the diurnal DO deficit profile times average water depth. Also shown are relationships between total daily respiration rate (R) and (C) the maximum absolute value of the diurnal DO deficit profile, and (D) the maximum absolute value of the diurnal DO deficit profile times average water depth. Data points are means for individual stations (n ranges from 3 to 11 dates per station) and error bars indicate one standard deviation. Pearson correlation coefficients and P values are indicated on each plot.

Figure 4. Relationships between catchment disturbance level and (A) average amplitude of the diurnal dissolved oxygen (DO), (B) average amplitude of the diurnal DO deficit times average water depth, (C) average maximum absolute value of the diurnal DO deficit, and (D) average maximum absolute value of the diurnal DO deficit times average



water depth. Lines of best fit of the data in each panel to an exponential model ( $Y = Y_0 - A \times \ln X$ ) are shown with BC1 excluded. This model explained 26% of the variation in panel A, 76% of the variation in panel B, 80% of the variation in panel C and 77% of the variation in panel D.







Figure 1.

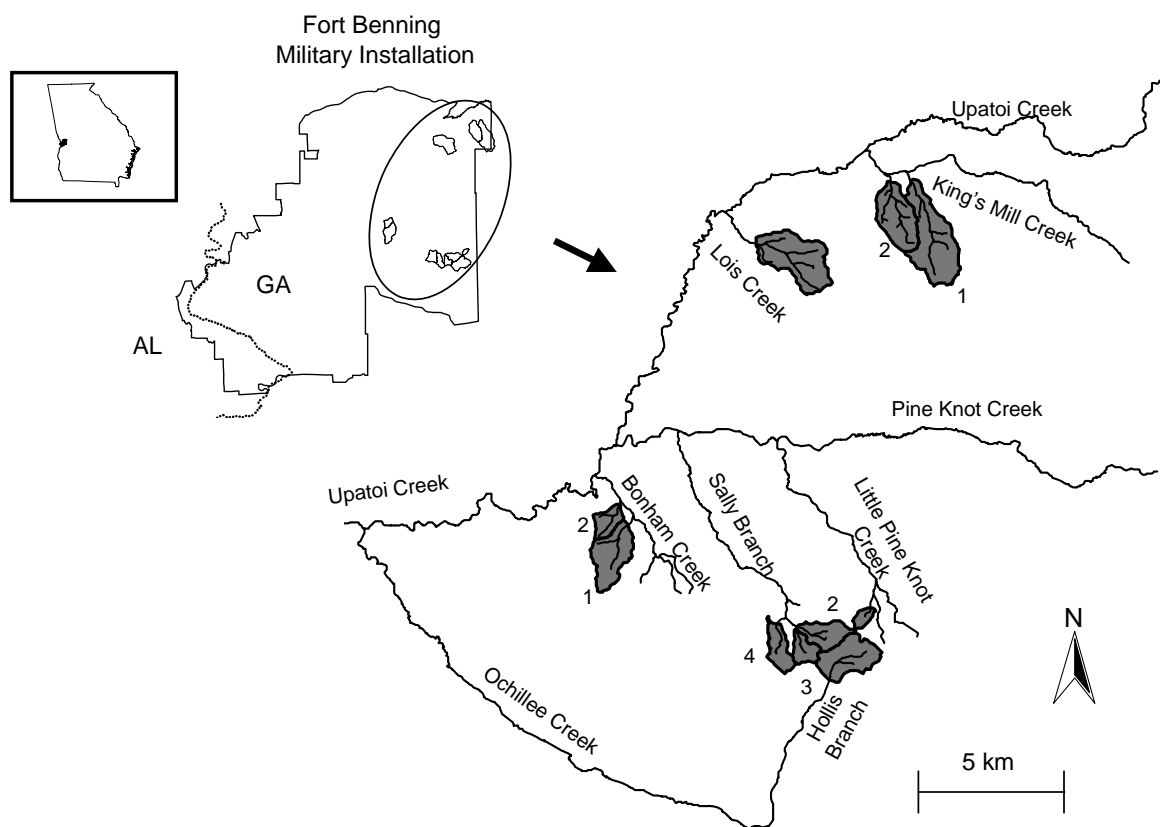




Figure 2.

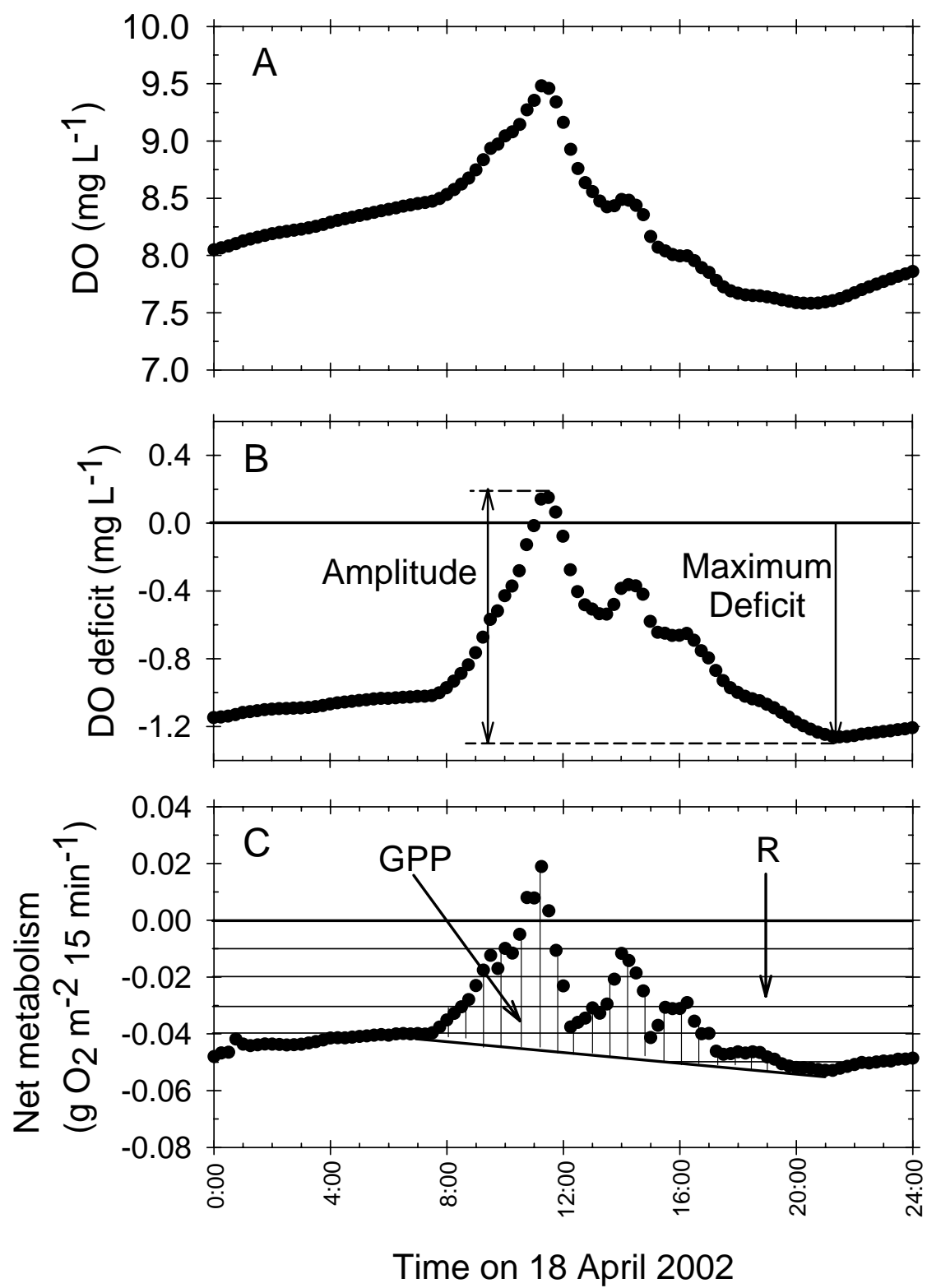








Figure 3.

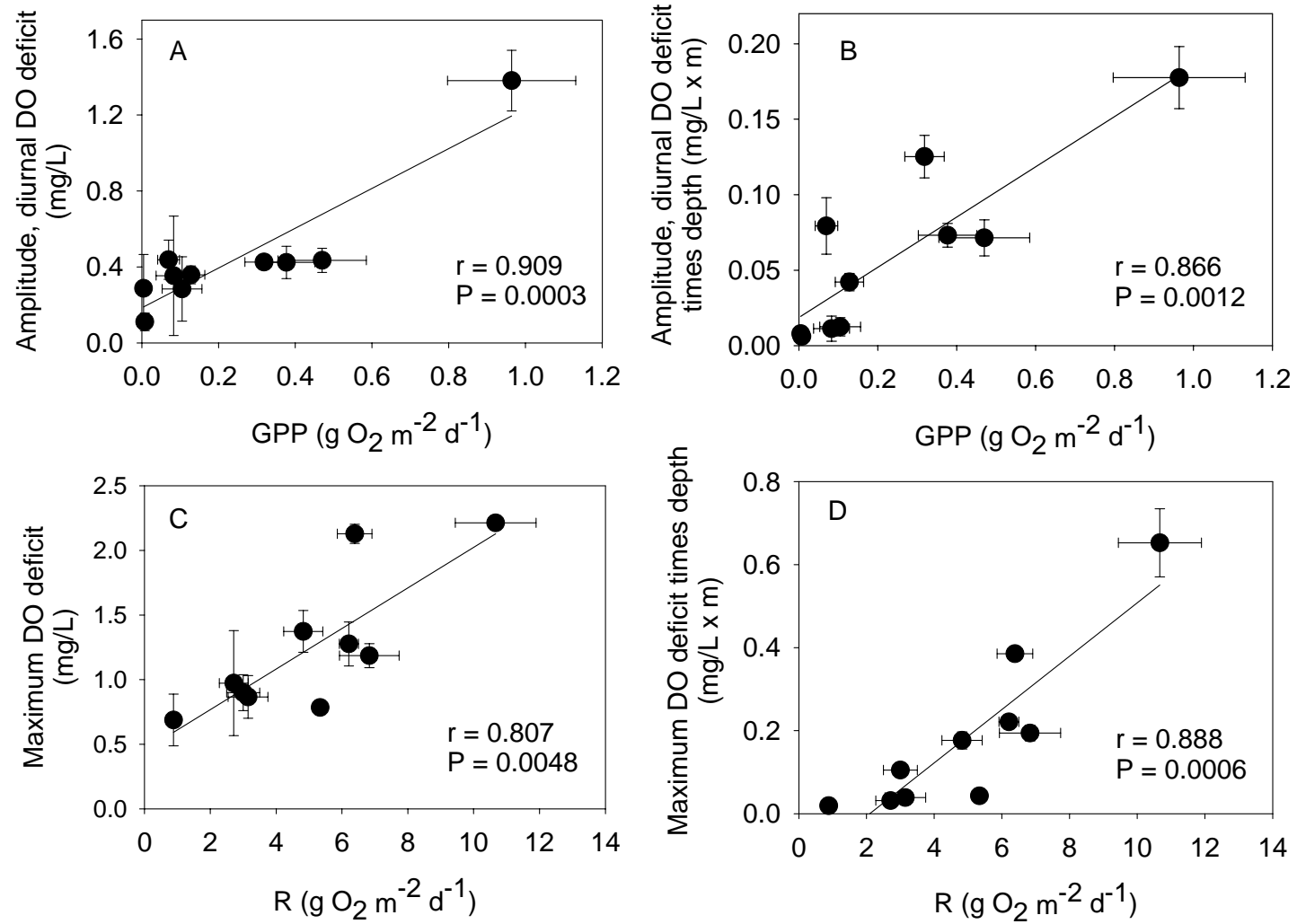
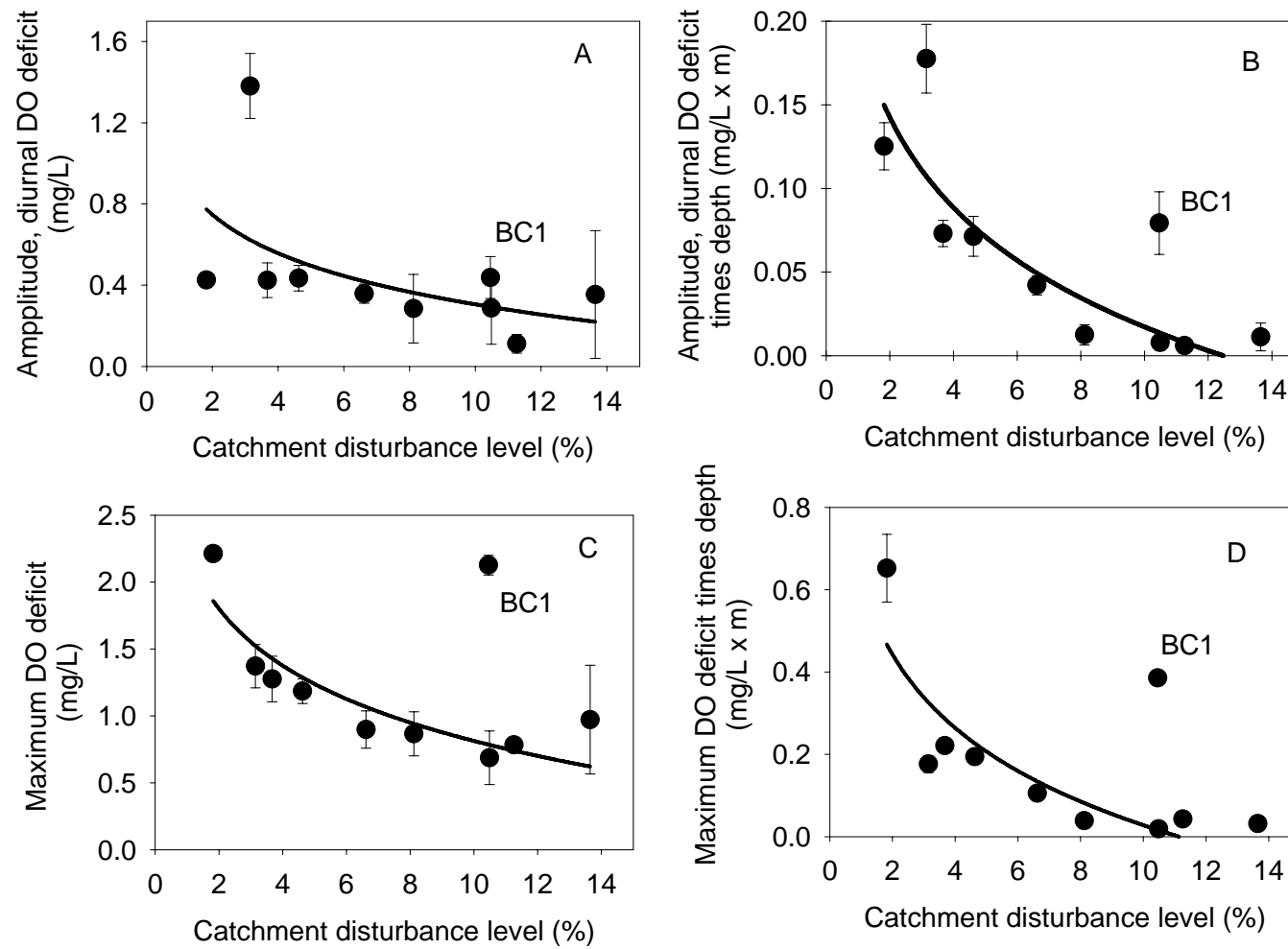




Figure 4.





Appendix D<sup>7</sup> of Indicators Final Report

**Evaluation of single- and multi-metric benthic macroinvertebrate indicators  
of catchment disturbance at the Fort Benning Military Installation, Georgia,  
USA**

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## Abstract

Stream benthic macroinvertebrates are useful indicators of catchment disturbance because they integrate catchment-scale ecological processes. We tested the ability of macroinvertebrate assemblages to indicate disturbance from military training at the Fort Benning Military Installation, Georgia, where the main disturbance to streams is influx of sediment associated with military training and use of unpaved roads. We studied seven small streams that drained catchments spanning a range of disturbance (measured as % of catchment as bare ground on slopes > 3% and under unpaved road cover). Nonmetric multidimensional scaling ordinations revealed macroinvertebrate assemblages were associated with catchment disturbance. Irrespective of season, several richness measures (e.g., number of Ephemeroptera, Plecoptera, and Trichoptera taxa and richness of Chironomidae) negatively corresponded with catchment disturbance, however except for chironomid richness all measures showed high variation among seasons and annually. Compositional and functional feeding group measures also showed high seasonal and annual variation, with only the % of macroinvertebrates clinging to benthic habitats (= % clingers) corresponding with disturbance. Both tolerance metrics tested, the Florida Index and North Carolina Biotic Index, showed little seasonal and annual variation, however only the Florida Index related to disturbance. A regional multimetric, the Georgia Stream Condition Index, consistently corresponded with catchment disturbance and showed the least temporal variability. Our results further suggest a threshold at 8 to 10% of the catchment as bare ground and unpaved road cover, a disturbance threshold similar to that reported for other land uses.



KEYWORDS: stream assessment, military training, disturbance, macroinvertebrates, metric, seasonal and annual variability.



## 1. Introduction

30 Streams are excellent systems to identify and test potential indicators of land use because they are intimately linked to their catchments, and thus integrate catchment-scale ecological processes and cumulative responses to disturbance. Benthic macroinvertebrates are a particularly useful indicator in this regard because they occur in a wide range of stream habitats, include a wide range of functional feeding groups, 35 compose the bulk of animal diversity, show a wide sensitivity range, have sedentary life cycles allowing relatively easy quantification, and have relatively short life cycles (usually <2 y), thus exposing most species to the source of impairment for most of their life cycle (Rosenberg and Resh, 1993).

A major impact of disturbance on stream ecosystems associated with catchment 40 land use is from increased sedimentation and concomitant decreased habitat availability, bed stability, and benthic organic matter in the channel (Chutter, 1969; Ryan, 1991; Quist et al., 2003; Maloney et al., In press). Macroinvertebrate response to sedimentation ranges from altered behavior (e.g., emigration from drift) to mortality (reviewed by Newcombe and MacDonald, 1991; Waters, 1995). Increased 45 sedimentation has also reportedly reduced macroinvertebrate density, biomass, and diversity (Lenat et al., 1981; Newcombe and MacDonald, 1991; Angradi, 1999), especially of sensitive taxa (Culp et al., 1986).

Response of macroinvertebrate assemblages to anthropogenic disturbance is often evaluated using “metrics”, which describe biological conditions from structural 50 and/or functional assemblage measures (Karr, 1991; Barbour et al., 1996,1999). Whereas single metrics reflect only 1 aspect of the assemblage (e.g., number of



Ephemeroptera taxa, Shannon's  $H'$ ) and may not indicate effects of multiple stressors, a multimetric incorporates several single assemblage/habitat metrics that encompass multiple aspects of assemblages and thus may provide a more powerful means of assessment (Karr et al., 1986).

Military installations provide excellent landscapes to examine the influence of catchment-scale disturbance on streams because they contain a wide range of disturbance conditions on relatively homogenous soils and vegetation. Numerous areas of installations have been undisturbed for over 50 years and serve as contemporary sources of high quality habitat (Cohn, 1996), whereas other areas often adjacent to or within a few kilometers of these undisturbed sites have been heavily disturbed (e.g., soil and vegetation disturbance) from military training repeatedly over decades.

Disturbance from training may reduce terrestrial vegetation, increase soil compaction, and reduce soil organic matter as well as alter terrestrial biodiversity (Severinghaus et al., 1981; Goran et al., 1983; Garten et al., 2003; Quist et al., 2003). Receiving streams in high military use catchments may be similarly disturbed, often by increased sedimentation associated with high amounts of bare ground resulting from training (Quist et al., 2003).

Numerous studies have assessed seasonal, spatial, and site variation in biotic metrics (e.g., Linke et al., 1999; Li et al., 2001; Clarke et al., 2002). However, to our knowledge, there have been no comparisons of variability between single metric and multimetric variability between seasons and years. Our objectives were to evaluate the efficacy of single- and multi-metric benthic macroinvertebrate measures to indicate catchment-scale disturbance from military land use, and assess the variability of both sets of metrics within sampling years and over years.



## 2. Methods

### Study site

80           Our study was located on the Fort Benning Military Installation (FBMI), in the  
Southeastern Plains ecoregion of central western Georgia, USA (Fig. 1). Fort Benning  
comprises ~735 km<sup>2</sup> and is primarily drained by Upatoi Creek. The climate is humid  
and mild with precipitation occurring year round (mean = 105 cm/y). Upland vegetation  
is mostly pine (*Pinus palustris*, *P. echinata*, and *P. taeda*) and mixed hardwoods (mostly  
85   *Quercus* spp.). Floodplain vegetation is primarily mesic hardwoods dominated by  
*Nyssa sylvatica*. Streams channels are low gradient (slopes 1–2.5%), sinuous, and  
sandy bottomed, often with large amounts of entrained coarse woody debris (CWD) and  
leaf litter (Felley, 1992, Maloney et al., In press).

          The US military purchased most FBMI land between 1918 and 1942, which  
90   became the home of the U.S. Army Infantry School. Collectively, FBMI hosts a wide  
range of military training activities including dismounted infantry, tracked vehicle  
maneuvers (i.e., tanks), heavy weapons usage, and airborne training drop zones  
(USAIC, 2001; Dale et al., 2002). These strongly contrasting land uses often are  
spatially segregated in different training compartments, making it possible to select  
95   study catchments that varied in the type of military activity and associated disturbance.

          We selected 7 catchments that spanned the range of landscape disturbance at  
FBMI. Our low-disturbance streams drained catchments mostly within compartments  
reserved for dismounted infantry training, whereas most high-disturbance streams were  
within compartments reserved for heavy-tracked vehicle training.



### Land use quantification

The most obvious disturbances from military land use at FBMI are the creation of bare ground from heavy-tracked vehicle training and creation and repeated use of unpaved roads and trails by military and nonmilitary vehicles (Maloney et al., In press).

105 These 2 disturbance sources were similar on remotely sensed data (at the catchment scale), so we could not ascertain their separate influences on stream invertebrate assemblages. Therefore, we quantified disturbance as the amount of bare ground on slopes >3% and unpaved road and trail cover in each catchment (%BGRD) using geographic information system (GIS) data sets (i.e., roads: 10-m resolution, 1995  
110 coverage; digital orthophotographs 1:5,000, July 1999; digital elevation models, DEMs, 1:24,000, 10-m grid size, 1993; and Landsat imagery, 28.5 m resolution, July and December 1999). We processed GIS data sets to catchment boundaries using ArcView© software (Environmental Systems Research Institute, Inc., Redlands, California, see also Maloney et al., In press).

115

### Macroinvertebrate sampling

We quantified benthic macroinvertebrates in May (spring), September (summer), and January (winter) over a 3-y period (January 2000 to September 2002), using 4 Hester-Dendy (HD) multiplate sampling units (Merritt and Cummins, 1996), with 3  
120 multiplates per unit, (total area = 0.33 m<sup>2</sup>). We left HDs in situ for a 6- to 8-wk colonization period. In addition, we also used 2 D-frame sweep net samples (250 µm) to sample general benthic habitats (e.g., runs) in each study site, with net samples taken downstream and upstream of HD stations on each date. We field-preserved all samples



in 95% EtOH, following elutriation of excess sediment. In the laboratory, we sorted the  
entire HD sample and subsampled sweep net samples (at least 200 organisms, see  
Vinson and Hawkins, 1996). We identified macroinvertebrates to the lowest taxonomic  
level possible (usually genus or morphospecies) using keys in Merritt and Cummins,  
1996), Wiggins, (1996), and Epler, (2001).

### Quantification of habitat

We characterized stream habitat both at the catchment and stream-reach scale.  
We quantified catchment area as the drainage area upgradient from our downstream-  
most sampling point using DEMs and ArcView. We measured stream discharge  
(incremental method, Gore, 1996) every ~2 mo using a Marsh-McBirney (model 2000)  
flow meter. We quantified stream temperature continuously using HOBO® H8  
temperature loggers (15- to 30-min intervals, Onset Computer Corporation, Bourne,  
Massachusetts) over the study in each stream. We calculated seasonal means for  
discharge and temperature. Stream site-specific current velocity and depth were based  
on means of 3 to 4 measures taken at each macroinvertebrate sampling location. We  
also visually estimated the relative abundance of instream CWD by classifying CWD  
over the streambed cover into 4 categories (0 = 0–25% of bed covered by CWD; 1 =  
26–50%, 2 = 51–75%; 3 = > 75%) to estimate the amount of local available woody  
habitat. Site-specific measures of current velocity, depth, and CWD relative abundance  
were averaged for each stream and then averaged by season.



## Metrics

We tested a variety of single metrics selected from standard EPA rapid  
150 bioassessment protocols (Barbour et al., 1999), two regionally defined tolerance  
metrics, the Florida Index (FLDEP, 2002) and the North Carolina Biotic Index (NCBI,  
NCDENR, 2003), and also a regional multimetric designed for Georgia streams  
(hereafter the Georgia Stream Condition Index, GASCI, GADNR, 2002). In total, we  
tested 9 richness, 10 composition, 5 feeding guild, and 2 tolerance single metrics and 1  
155 multimetric (Table 1).

For the Florida index, we separated taxa into 3 classes, with class 1 (sensitive)  
being assigned a value of 2, class 2 (moderately tolerant) assigned a value of 1, and  
class 3 (tolerant) a value of 0. The index is the sum of taxa in the respective classes,  
with lower and higher values indicating greater and lesser likelihood of stream  
160 impairment, respectively (FLDEP, 2002). NCBI is a tolerance index derived for streams  
in North Carolina. Taxa are assigned tolerance values according to tolerance level,  
based on a 0–10 scale, where 0 represents intolerant taxa and 10 represents highly  
tolerant taxa (NCDENR, 2003). NCBI is calculated as:

$$NCBI = \left( \frac{\sum(TV_i)(n_i)}{N} \right) \quad \text{equation 1,}$$

165 where  $TV_i$  is the tolerance value of taxa  $i$ ,  $n_i$  is the abundance value of taxa  $i$ , and  
 $N$  = sum of abundance values for all taxa in the sample (NCDENR, 2003). NCBI scores  
range from 0–10, where 0 indicates the best water quality and 10 indicates the worst.  
As specified in the NCDENR manual, we used a correction factor for non-summer  
samples for the Coastal Plains ecoregions (NCDENR 2003).



170           The GASCI incorporates 7 macroinvertebrate metrics (taxa richness, EPT richness, Chironomidae richness, % dominant taxa, % Diptera, Florida Index, % Filterers) and 1 habitat metric (calculated by summation of assessment scores of 10 habitat characteristics, including channel sinuosity, bank stability, riparian vegetation features, see GADNR 2002). Each metric is assigned a score of 1, 3, or 5 depending  
175   on predefined ranges for each metric for the summer and winter seasons. For example, taxa richness has the following pre-defined ranges and scores: >30 taxa given a score of 5, 16–30 taxa score = 3, and <16 taxa score = 1. Each metric has predefined ranges with associated scores and the overall GASCI score is the summation of individual metric scores for each stream.

#### 180   Data analysis

          We used Nonmetric Multidimensional Scaling ordinations (NMS, McCune and Grace, 2002) to examine seasonal macroinvertebrate assemblage similarity within and among streams. Nonmetric Multidimensional Scaling is an indirect gradient analysis  
185   technique that uses pairwise dissimilarity matrices to estimate site (stream) position in species space (Jongman et al., 1995), and is a more robust ordination method than principal components analysis or detrended correspondence analysis (Minchin, 1987). Axes scores from NMS can be related to environmental variables to reveal ecological patterns (Hawkins et al., 1997). Because rare taxa provide redundant information for  
190   ordinations and the final stress of an NMS ordination increases with sample size (Marchant, 1999; McCune and Grace, 2002) we removed rare taxa (<10% of samples) prior to ordinations. We also square-root transformed abundance data prior to ordinations using PC-ORD (MjM Software Design, Gleneden Beach, Oregon). We then



regressed stream-specific NMS scores with independent variables to determine which  
195 habitat or land use variable was related to macroinvertebrate assemblages.

We also used correlation analysis on season-average stream metric values to  
assess the degree to which macroinvertebrate metrics were related to catchment  
disturbance (as %BGRD). We transformed all indicator variables (counts – arcsine  
square root, percentages – square root) as necessary prior to analyses to satisfy  
200 normality. Our repeated sampling design (2-3 years, 3 seasons per stream) allowed us  
to further assess the variability of each metric both among seasons and years. For  
seasonal variation (i.e. variability within 1 year over different seasons) we calculated  
and compared coefficient of variation (CV) for each year for each stream, using season  
values as replicates ( $n = 3$ ). For annual variation, we calculated and compared season-  
205 specific CVs for each stream using annual values as replicates ( $n = 2-3$  depending on  
stream).

### 3. Results

Study catchments were small (range = 0.33 to 5.43 km<sup>2</sup>) and showed a wide  
210 range of landscape disturbance (%BGRD, 3.15 to 14.66%, Table 2). Average stream  
discharge and current velocity ranged from <0.001 to 0.045 m<sup>3</sup>/s and 0.028 to 0.289  
m/s, respectively. Average stream width and water depth ranged 0.75 to 2.44 m and  
0.02 to 0.19 m, respectively (Table 2). Site-specific CWD cover ranged from 0.65 (<  
25% of the streambed) to 2.58 (51-75% cover). Maximum and average temperatures  
215 were highest in summer (28.7°C, 22.3°C, respectively), followed by spring (24.5°C,  
16.1°C) and winter (18.62 °C, 8.4°C).



Nonmetric Multidimensional Scaling using macroinvertebrates successfully distinguished variation in catchment disturbance among study streams, but also suggested an influence of channel and seasonal variables on assemblages (Fig. 2).

Axis 1 accounted for most of the variation in assemblages ( $R^2 = 0.46$ ), followed by axis 3 ( $R^2 = 0.23$ ), and axis 2 ( $R^2 = 0.14$ ). When axes 1 and 3 were regressed against land use and habitat variables Axis 1 was best explained by catchment disturbance (as %BGRD) ( $R^2 = 0.77$ ,  $P < 0.001$ ), whereas Axis 3 corresponded to average streamwater temperature and depth ( $R^2 = 0.43$ ,  $P = 0.007$ ). These relationships suggest that assemblage similarity associated with axis 1 was driven by land use, whereas assemblage similarity associated with axis 3 was driven by habitat and season.

Correlation analysis revealed that macroinvertebrate richness measures were the best simple metrics indicating disturbance from land use. The number of clinger taxa consistently related to catchment disturbance (as %BGRD, Table 3). Total number of EPT taxa ranged from 2 to 16 per stream per season and was negatively related to %BGRD in all seasons (Fig. 3, Table 3). At %BGRD levels >10 EPT richness fell below the 95% confidence limit for the 3 least-disturbed streams (Fig. 3), which was used as a measure of low-disturbance variation. The number of Ephemeroptera taxa consistently correlated with %BGRD as did the number of Trichoptera taxa during spring and winter, whereas the number of Plecoptera taxa (range from 0–6 per season) corresponded with %BGRD only during spring (Table 3). The number of Chironomidae taxa was strongly inversely correlated with catchment disturbance in all seasons and consistently showed values below the 95% confidence limit of the 3 least-disturbed streams for catchments with BGRD values > 10% (Table 3, Fig. 3). The number of *Tanytarsini* taxa, a tribe of Chironomidae, was consistently negatively related to %BGRD.



Composition and feeding measures typically showed no relationship with %BGRD (Table 3). Only %clingers consistently indicated catchment disturbance, being negatively related to %BGRD in all seasons (Table 3). Tolerance metrics showed mixed success as indicators of catchment disturbance. The Florida Index scores were negatively related to %BGRD in all seasons (Fig. 3). At %BGRD levels >10 the Florida Index fell consistently below the 95% confidence limit for the 3 least-disturbed streams (Fig. 3). The NCBI showed no relationship to %BGRD in spring and summer and only a weak relationship in winter (Table 3). Regardless of season, the GASCI was negatively correlated with %BGRD (Table 3). At BGRD > 8% GASCI scores fell below the 95% confidence limit for the 3 least-disturbed streams (Fig. 4).

Variability analysis revealed that macroinvertebrate composition and functional feeding group metrics were more variable annually (mean CV = 52.0 and 43.4%, respectively, Fig. 5) and seasonally (38.6 and 37.1%, respectively) than richness metrics (25.0 and 20.9%, respectively, Fig. 5) and tolerance metrics (7.6 and 7.9%, respectively). The GASCI multimetric index showed lower seasonal and annual variability than the other metrics (mean CV = 3.7%, 6.3%, respectively, Fig. 5).

#### **4. Discussion**

A useful ecological indicator of disturbance should be easily measured, sensitive, anticipatory, and integrative across key environmental gradients (Cairns et al., 1993; Dale and Beyeler, 2001). Land managers also require indicators that have low variation among seasons and/or years. Results of our study suggest that stream benthic macroinvertebrate assemblages are useful indicators of catchment disturbance on military lands (i.e., our %BGRD measure). Our results also suggest that, excluding



265 Chironomidae richness, all single metrics showed high seasonal and annual variation,  
so they appeared less useful than the substantially less temporally variable Florida  
Index and the multimetric GASCI. Our results further suggest a disturbance threshold  
between 8–10% of the catchment as bare ground and unpaved road cover where  
several metrics fell below the 95% confidence limit for the 3 least disturbed streams.  
270 Interestingly, although catchment disturbance from military training is different than that  
from urbanization (e.g., impervious surface), this threshold is within the range of  
disturbance thresholds identified for urbanized impacts on streams (8–20%, Arnold and  
Gibbons, 1996; Paul and Meyer, 2001; Wang et al., 2001).

The main disturbance in our catchments was removal of vegetation, concomitant  
275 disruption of soil surface, and general use of unpaved roads, each of which likely  
increased erosion and subsequent sedimentation of stream channels. Results of our  
NMS ordination suggested the macroinvertebrate assemblage, as a whole, was affected  
by catchment disturbance, as indicated by the high separation of site scores along axis  
1, the axis that was highly associated with %BGRD. A likely driving force in separation  
280 among benthic assemblages in these streams was a reduction of available in-stream  
habitat (CWD) and decreased bed stability in high-disturbed streams (Maloney et al., In  
press). Amounts of CWD (and associated snags) and degree of bed stability have been  
shown to affect available macroinvertebrates habitat in sandy-bottomed streams (Benke  
et al., 1984; Wallace and Benke, 1984; Benke and Wallace, 1990). Moreover, the NMS  
285 analysis revealed a seasonal influence on macroinvertebrate assemblages, with  
summer NMS scores being lower (i.e., assemblages more dissimilar) than respective  
spring or winter scores in most streams. Summer assemblages were possibly  
influenced by physical stress as evident from summer showing the lowest average



discharge and site-specific flow, depth, and width of the 3 seasons in all seasons (Table  
290 2). Strong seasonality in benthic macroinvertebrates have been reported from other  
sandy-bottomed streams (Benke et al., 1984; Lenat, 1993) possibly in response to  
hydrologic stress (Felley, 1992, but see Boulton et al., 1992; Hutchens et al., 1998).

Except for Chironomidae richness, our data suggest that use of single metrics did  
not adequately indicate catchment disturbance at FBMI as a result of high seasonal and  
295 annual variation, low correlation with disturbance, or a combination of both high  
variation and low correspondence with disturbance. For example, EPT richness  
correlated well with disturbance in all seasons and had sufficient number of taxa  
seasonally collected per stream (4–15); however it also showed high seasonal and  
annual variation. The number of EPT taxa is a widely used indicator of stream integrity  
300 (Plafkin et al., 1989; Klemm et al., 2002) and is incorporated within numerous  
multimetric indices (Barbour et al., 1999; Royer et al., 2001; Ofenböck et al., 2004). Our  
data suggest its high variability may preclude its use as a stand-alone metric in low  
gradient, sandy-bottomed streams. Further, it is important to note that the separate use  
of Ephemeroptera, Plecoptera, and Trichoptera taxa as indicators was inadequate  
305 because of the low numbers of these taxa seasonally collected per stream  
(Ephemeroptera 0–4, Plecoptera 0–6, and Trichoptera 1–8, K. O. Maloney unpublished  
data) and their relatively high seasonal and annual variation, which together resulted in  
low detection ability. Low numbers of taxa in these aquatic insect orders has been  
reported elsewhere for sandy-bottomed streams (Barbour et al., 1996) and may  
310 generally constrain the separate use of these metrics in such streams. Moreover,  
composition and feeding guild metrics were not reliable indicators of disturbance and  
generally had high seasonal variation as well as high annual variation. The only



successful indicator within this group, the % of clinger taxa, consistently showed a negative relationship with disturbance likely in response to lower amounts of CWD in high-disturbance catchments (Maloney et al., In press). These taxa require firm substrate such as CWD as habitat. The % of clinger taxa, however, showed high seasonal and annual variation suggesting an inconsistent temporal relationship with catchment disturbance. The high variation of composition and feeding group metrics has been reported elsewhere (Resh and Jackson, 1993; Fore et al., 1996) and may be a result of the inherent patchy nature of stream substrates (Pringle et al., 1988; Townsend, 1989) as well as the difficulty in accurately assigning species to specific feeding groups (Cummins and Klug, 1979; Merritt and Cummins, 1996).

The number of Chironomidae taxa was a reliable indicator of disturbance and had relatively low seasonal and annual variation. We collected a total of 50 chironomid taxa (>25% of the total taxa collected), which likely vary greatly in trophic level, habitat preference, life cycle, and tolerance (Beck, 1954; Merritt and Cummins, 1996; Vuori and Joensuu, 1996; Barbour et al., 1999; Shaw and Richardson, 2001). Such features make Chironomidae a good candidate for assessing stream health, especially in streams where they are diverse and numerically dominant. Chironomids compose the bulk of macroinvertebrate richness in many streams in North America (Barton, 1980; Benke et al., 1984; Jackson and Fisher, 1986; Bourassa and Morin, 1995), South America (Bojsen and Jacobsen, 2003; Miserendino, 2004), Europe (Aagaard et al., 1997; Solimini et al., 2001; Arscott et al., 2003) and Australia (Kay, 2001; Milner et al., 2001). As a cautionary note, however, use of subsets of Chironomidae was not as effective in indicating disturbance in our study. For example, the number of *Tanytarsini* taxa was negatively related to %BGRD in all seasons and also showed low seasonal



and annual variation, however this metric was not robust because of low richness in our study (2–4 taxa per stream per season).

Regionally defined tolerance metrics and multimetric indices showed mixed ability to indicate catchment disturbance. The NCBI was unrelated to catchment disturbance in spring and summer and only weakly related to disturbance in winter, whereas the Florida Index successfully indicated catchment disturbance in all seasons. The Florida index also had relatively low seasonal and annual variation. However, the Florida Index uses a presence/absence approach, which may not identify alterations to the assemblage compositional structure from disturbance. The GASCI incorporates the Florida Index along with 6 other metrics (2 of which, number of EPT taxa and number of Chironomidae taxa were useful indicators) and a habitat assessment score into a single robust score, reducing the limitations of a presence/absence method. The GASCI indicated catchment disturbance for both the summer and winter and had relatively low seasonal and annual variation, which taken together suggest the GASCI was a highly successful indicator for our streams.

Land managers must be equipped with the means to accurately and consistently assess potential impacts of land use on receiving streams. Our results suggest land use managers at Fort Benning as well as those responsible for conservation of the ~30 million acres of land at ~6000 locations managed by the US Department of Defense (USDoD, 2004), especially those in the Southeast US (e.g., Fort Stewart, Fort Mitchell, Fort Polk), may be able to use pre-existing, regional, multimetrics to ascertain the degree of stream impairment caused by training and related land uses. Moreover, our results provide support for the highly variable nature of single metrics (e.g., Linke et al., 1999; Li et al., 2001; Clarke et al., 2002). Our results expand on these earlier findings



by reporting the high variation in single metrics not only among seasons but also years.

Most states in the US have programs already in place that incorporate multimetric

approaches to bioassessment (Barbour et al., 2000), however some states still rely

heavily on use of single metrics (Klemm et al., 2002). The high seasonal and annual

365 variation of single metrics observed in our study may, during times when such metrics

are used alone, result in misidentification of impaired conditions and/or site

misclassification. It is logical to infer, therefore, that regionally defined multimetrics such

as those developed within the US and Europe may be the most appropriate and

accurate indicators of stream integrity (Barbour et al., 1999; Vlek et al., 2004).

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Table 1. Definitions and predicted response of macroinvertebrate metrics used in the study. Each metric's predicted response to disturbance came from published literature (Cummins et al., 1989; Kerans and Karr, 1994; Barbour et al., 1996; Barbour et al., 1999). EPT = Ephemeroptera (mayflies), Plecoptera (stoneflies), Trichoptera (caddisflies); SCI = Stream Condition Index, FFG = functional feeding group. \* indicates single metrics used in GASCI calculation. See text for information about GASCI.

| Measure       | Metric                            | Definition  | Predicted response |
|---------------|-----------------------------------|---|--------------------|
| Richness      | No. of Ephemeroptera taxa         | Number of taxa in the order Ephemeroptera                           | Decrease           |
|               | No. of Plecoptera taxa            | Number of taxa in the order Plecoptera                              | Decrease           |
|               | No. of Trichoptera taxa           | Number of taxa in the order Trichoptera                             | Decrease           |
|               | No. of EPT taxa*                  | Number of taxa in the orders Ephemeroptera, Plecoptera, Trichoptera | Decrease           |
|               | No. of Chironomidae taxa*         | Number of taxa in the family Chironomidae                           | Decrease           |
|               | No. of Orthoclaadiinae taxa       | Number of taxa in the subfamily Orthoclaadiinae                     | Decrease           |
|               | No. of Tanytarsini taxa           | Number of taxa in the tribe Tanytarsini                             | Decrease           |
|               | No. of taxa*                      | Number of aquatic insect taxa                                       | Decrease           |
|               | No. of clinger taxa               | Number of clinger habitat group                                     | Decrease           |
| Composition   | Shannon's $H'$                    | Species diversity of assemblage                                     | Decrease           |
|               | % Diptera*                        | % of assemblage as dipterans  | Increase           |
|               | % Ephemeroptera                   | % of assemblage as mayflies   | Decrease           |
|               | % Plecoptera                      | % of assemblage as stoneflies                                       | Decrease           |
|               | % Trichoptera                     | % of assemblage as caddisflies                                      | Decrease           |
|               | % Hydropsychidae of Trichoptera   | % of total caddisflies as Hydropsychidae                            | Increase           |
|               | % Oligochaeta                     | % of assemblage as aquatic worms                                    | Variable           |
|               | % Orthoclaadiinae of Chironomidae | % of chironomids as Orthoclaadiinae                                 | Increase           |
|               | % Dominant of total*              | Dominance of most abundant taxa                                     | Increase           |
|               | % Clingers                        | % of clinger habitat group  | Decrease           |
|               | % Tanytarsini of Chironomidae     | % of chironomids in the tribe Tanytarsini                           | Decrease           |
| Feeding guild | % Predators                       | % of FFG as predators   | Variable           |
|               | % Scrapers                        | % of FFG as scrapers  | Decrease           |
|               | % Shredders                       | % of FFG as shredders   | Decrease           |
|               | % Filterers*                      | % of FFG as collector-filterers                                     | Decrease           |
|               | % Collector - gatherers           | % of FFG as collector-gatherers                                     | Variable           |
| Tolerance     | Florida Index*                    | Weighted sum of intolerant taxa                                     | Decrease           |
|               | NCBI                              | North Carolina Biotic Index   | Decrease           |
| Multimetric   | GASCI                             | Georgia Stream Condition Index                                      | Decrease           |



Table 2. Study stream and catchment characteristics in order of %BGRD. %BGRD = the proportion of bare ground on slopes >3% and unpaved road cover in a catchment, CWD = coarse woody debris measured as the planar coverage of stream bottom, Min = minimum, Max = maximum, ND = no data, Site-specific indicates mean values for site locations in a stream.

| Stream                           | Abbreviation | %BGRD | Drainage<br>area (km <sup>2</sup> ) | Season | Discharge<br>(m <sup>3</sup> /s) | Site-specific    |              |              |  | Temperature (°C) |       |       |
|----------------------------------|--------------|-------|-------------------------------------|--------|----------------------------------|------------------|--------------|--------------|--|------------------|-------|-------|
|                                  |              |       |                                     |        |                                  | Current<br>(m/s) | Depth<br>(m) | Width<br>(m) | CWD<br>(m <sup>2</sup> /m <sup>2</sup> ) | Mean             | Min   | Max   |
| Bonham Creek<br>Tributary 2      | BC2          | 3.15  | 0.75                                | Spring | 0.003                            | 0.074            | 0.11         | 0.96         | 0.65                                     | 17.41            | 7.83  | 23.63 |
|                                  |              |       |                                     | Summer | 0.001                            | 0.028            | 0.06         | 0.75         | 1.04                                     | 22.89            | 17.14 | 31.12 |
|                                  |              |       |                                     | Winter | 0.008                            | 0.171            | 0.14         | 1.18         | 1.17                                     | 8.64             | 2.03  | 16.00 |
| Lois Creek                       | LC           | 3.67  | 3.32                                | Spring | 0.040                            | 0.289            | 0.19         | 1.34         | 1.13                                     | 15.64            | 9.00  | 26.30 |
|                                  |              |       |                                     | Summer | 0.008                            | 0.128            | 0.10         | 1.15         | 1.54                                     | 23.20            | 18.28 | 26.73 |
|                                  |              |       |                                     | Winter | 0.019                            | 0.189            | 0.15         | 1.64         | 1.77                                     | 8.32             | 2.46  | 26.70 |
| King's Mill Creek<br>Tributary 1 | KM1          | 4.63  | 3.69                                | Spring | 0.044                            | 0.139            | 0.19         | 2.13         | 0.80                                     | 16.57            | 8.23  | 23.63 |
|                                  |              |       |                                     | Summer | 0.014                            | 0.097            | 0.15         | 2.07         | 1.47                                     | 22.38            | 17.14 | 25.95 |
|                                  |              |       |                                     | Winter | 0.028                            | 0.130            | 0.17         | 2.24         | 1.45                                     | 8.84             | 2.03  | 26.73 |
| Sally Branch<br>Tributary 1      | SB1          | 8.38  | 5.43                                | Spring | 0.045                            | 0.243            | 0.09         | 1.90         | 0.79                                     | 15.82            | 8.63  | 29.90 |
|                                  |              |       |                                     | Summer | 0.007                            | 0.146            | 0.06         | 1.55         | 0.79                                     | 22.91            | 17.90 | 33.17 |
|                                  |              |       |                                     | Winter | 0.025                            | 0.222            | 0.17         | 2.44         | 1.33                                     | 6.91             | 0.29  | 14.85 |
| Bonham Creek<br>Tributary 1      | BC1          | 10.46 | 2.10                                | Spring | 0.008                            | 0.110            | 0.17         | 1.27         | 1.20                                     | 15.19            | 8.63  | 21.71 |
|                                  |              |       |                                     | Summer | 0.005                            | 0.103            | 0.13         | 1.17         | 1.63                                     | 21.87            | 18.28 | 29.90 |
|                                  |              |       |                                     | Winter | 0.012                            | 0.116            | 0.17         | 1.52         | 2.58                                     | 9.03             | 2.89  | 15.62 |
| Little Pine Knot                 | LPK          | 11.26 | 0.33                                | Spring | 0.003                            | 0.118            | 0.06         | 1.04         | 0.70                                     | 17.22            | 9.82  | 21.71 |
|                                  |              |       |                                     | Summer | 0.002                            | 0.111            | 0.05         | 0.83         | 0.83                                     | 21.42            | 17.52 | 28.70 |
|                                  |              |       |                                     | Winter | 0.003                            | 0.137            | 0.08         | 1.03         | 1.01                                     | 11.21            | 5.81  | 17.14 |
| Sally Branch<br>Tributary 5      | SB5          | 14.66 | 1.27                                | Spring | 0.014                            | 0.147            | 0.06         | 0.95         | 0.94                                     | 15.09            | 6.20  | 24.80 |
|                                  |              |       |                                     | Summer | <0.001                           | ND               | 0.02         | 0.79         | 1.00                                     | 22.53            | 19.04 | 25.17 |
|                                  |              |       |                                     | Winter | 0.002                            | 0.076            | 0.07         | 1.14         | 1.08                                     | 5.62             | -1.06 | 13.32 |



Table 3. Pearson correlation coefficients between benthic macroinvertebrate metrics and the proportion of catchment disturbance as bare ground and unpaved road cover (%BGRD). Metric abbreviations defined in Table 1. NA = metric not applicable in this season. \*= $P < 0.10$ , \*\*= $P < 0.05$ . '–' = nonsignificant ( $P > 0.05$ ).

| Type          | Metric                           | Spring  | Summer  | Winter  |
|---------------|----------------------------------|---------|---------|---------|
| Richness      | No. of Ephemeroptera taxa        | –0.81** | –0.82** | –0.88** |
|               | No. of Plecoptera taxa           | –0.74** | –       | –       |
|               | No. of Trichoptera taxa          | –0.81** | –       | –0.87** |
|               | No. of EPT taxa                  | –0.88** | –0.72*  | –0.93** |
|               | No. of Chironomidae taxa         | –0.76** | –0.83** | –0.84** |
|               | No. of Orthocladiinae taxa       | –       | –       | –       |
|               | No. of Tanytarsini taxa          | –0.85** | –0.77** | –0.90** |
|               | No. of taxa                      | –0.72*  | –       | –0.84** |
|               | No. of clinger taxa              | –0.91** | –0.73*  | –0.96** |
| Composition   | Shannon's $H'$                   | –       | –       | –       |
|               | % Diptera                        | –       | –       | –       |
|               | % Ephemeroptera                  | –       | –       | –0.77** |
|               | % Plecoptera                     | –       | –       | –       |
|               | % Trichoptera                    | –0.69*  | –       | –0.70*  |
|               | % Hydropsychidae of Trichoptera  | –       | 0.78**  | –       |
|               | % Oligochaeta                    | –       | –       | –       |
|               | % Orthocladiinae of Chironomidae | –       | –       | 0.71*   |
|               | % Dominant of total              | –       | 0.96**  | –       |
|               | % Clingers                       | –0.77** | –0.69*  | –0.79** |
| Feeding guild | % Tanytarsini of Chironomidae    | –       | –       | –0.69*  |
|               | % Predators                      | –       | –       | –       |
|               | % Scrapers                       | –       | –0.89** | –       |
|               | % Shredders                      | –       | –       | –       |
|               | % Filterers                      | –       | –       | –       |
| Tolerance     | % Collector - gatherers          | –       | –       | –       |
|               | Florida Index                    | –0.97** | –0.88** | –0.95** |
|               | NCBI                             | –       | –       | 0.71*   |
| Multimetric   | GASCI                            | NA      | –0.96** | –0.95** |



Figure legends:

Figure 1. Locations of study catchments (polygons) within Fort Benning Military Installation, GA. Dotted line in middle figure represents the Chattahoochee River, which separates Alabama (AL) and Georgia (GA). Numbers in the right figure identify watersheds on the same stream (e.g., 1 and 2 on the Bonham Creek represent Bonham Creek Tributaries 1 and 2, BC1 and BC2, respectively).

Figure 2. Results of nonmetric multidimensional scaling (NMS) ordination. Symbols represent stream-specific macroinvertebrate scores, and lines connecting symbols show seasonal changes in scores for each stream. Open circles = winter, shaded circles = spring, and solid circles = summer.  $R^2$  values represent the proportion of variation in the macroinvertebrate assemblage similarity accounted for by each axis. Arrows on axes indicate direction of relationships between habitat variables and NMS scores (see text for values). Axis scores are raw values, stress level = 11.8 for the 3-dimension solution, with a final instability of 0.00001 after 91 iterations.

Figure 3. Relationships between %BGRD (% of catchment as bare ground on slopes > 3% and unpaved road cover) and EPT richness, Chironomidae richness, and Florida Index by season. Solid lines represent trends using means (all trends significant at  $p = 0.05$  see Table 3 for  $r$  and  $p$  values), dashed lines represent 95% upper and lower confidence limits of the 3 least disturbed streams in the data set (BC2, KM1, K13). Points represent individual stream values over several years.



Figure 4. Relationships between Georgia Stream Condition Index (GASCI) values and %BGRD (% of catchment as bare ground on slopes > 3% and unpaved road cover) for summer and winter. Solid lines represent trends using means (all trends significant at  $p = 0.05$  see Table 3 for  $r$  and  $p$  values), dashed lines represent 95% upper and lower confidence limits of the 3 least disturbed streams in the data set (BC2, KM1, K13). Points represent individual stream values over several years. Numbers in parentheses indicate overlapping points.

Figure 5. Mean (+ 1 SE) coefficient of variation (CV) of the 5 categories of macroinvertebrate metrics among seasons (spring, summer, winter) and years (2000–2002) by category. Numbers in parentheses indicate the number of metrics in each category.



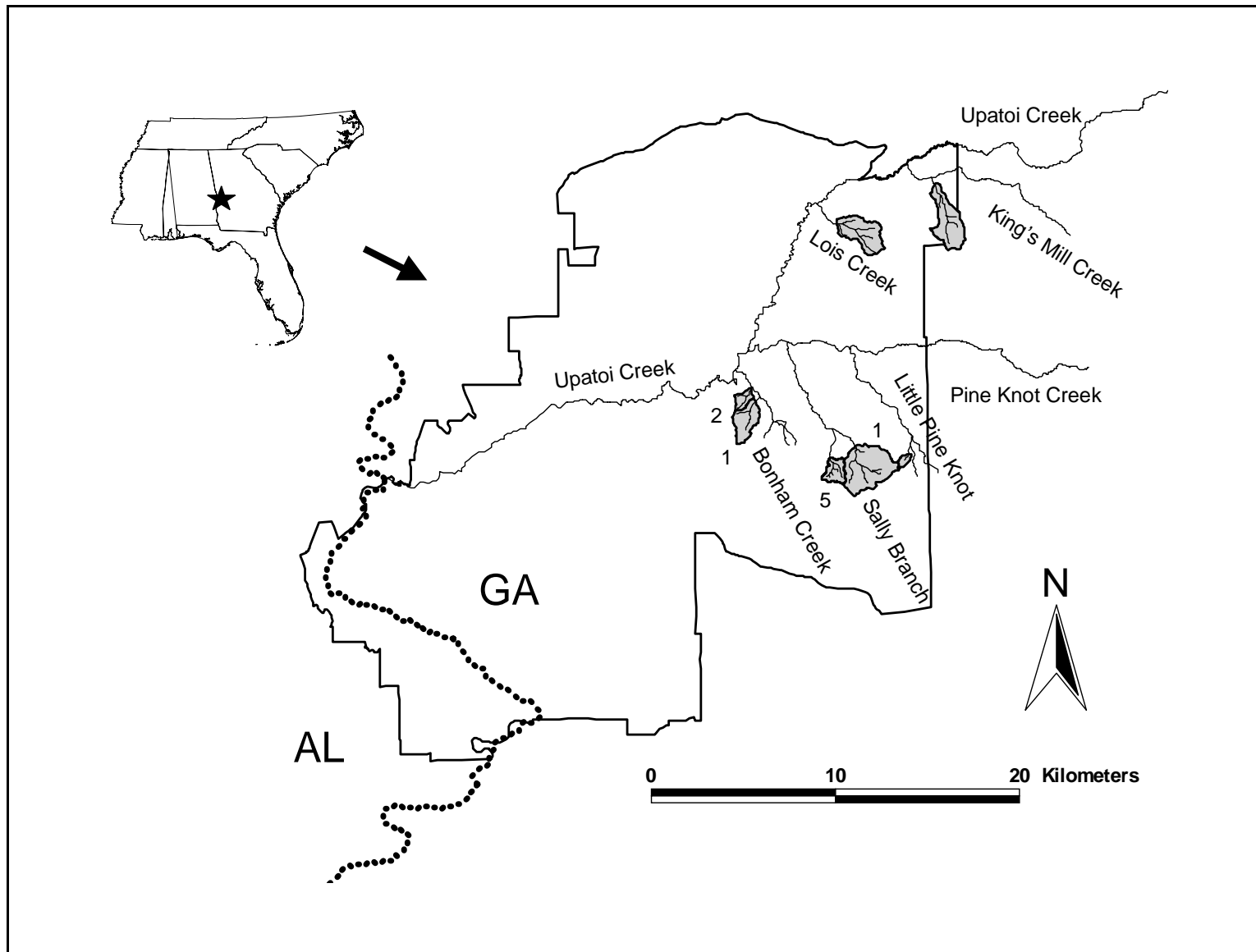


Figure 1.



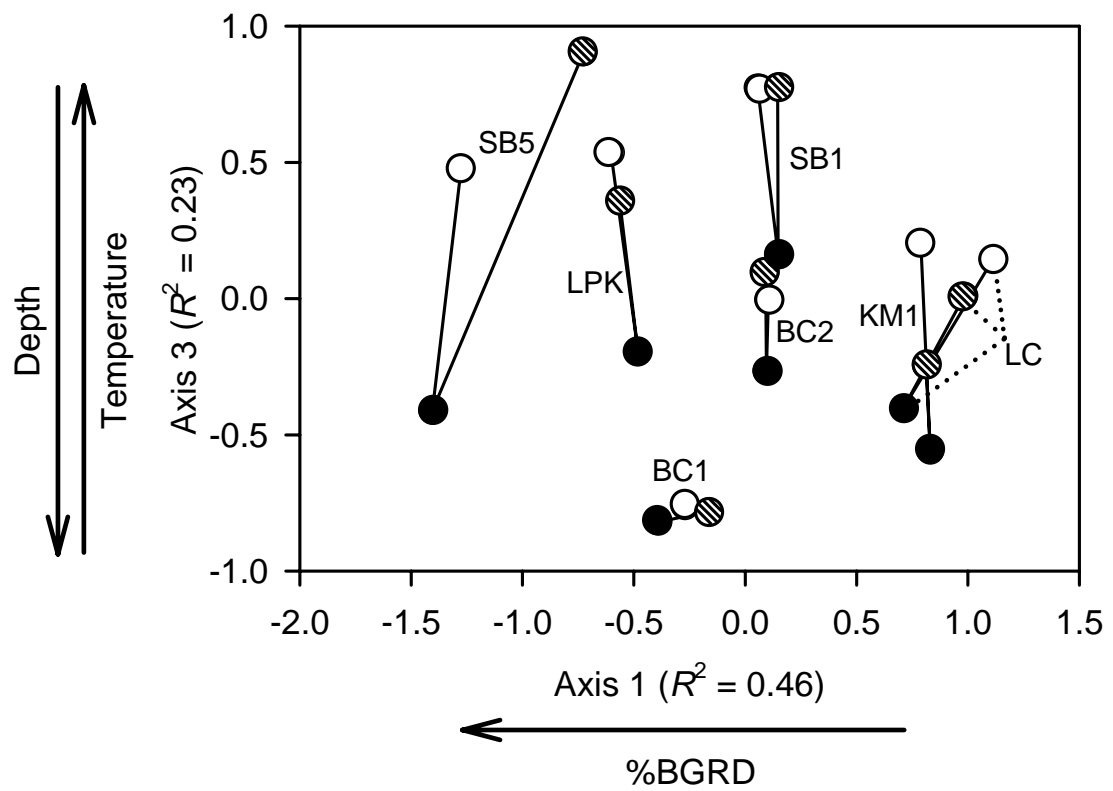


Figure 2.



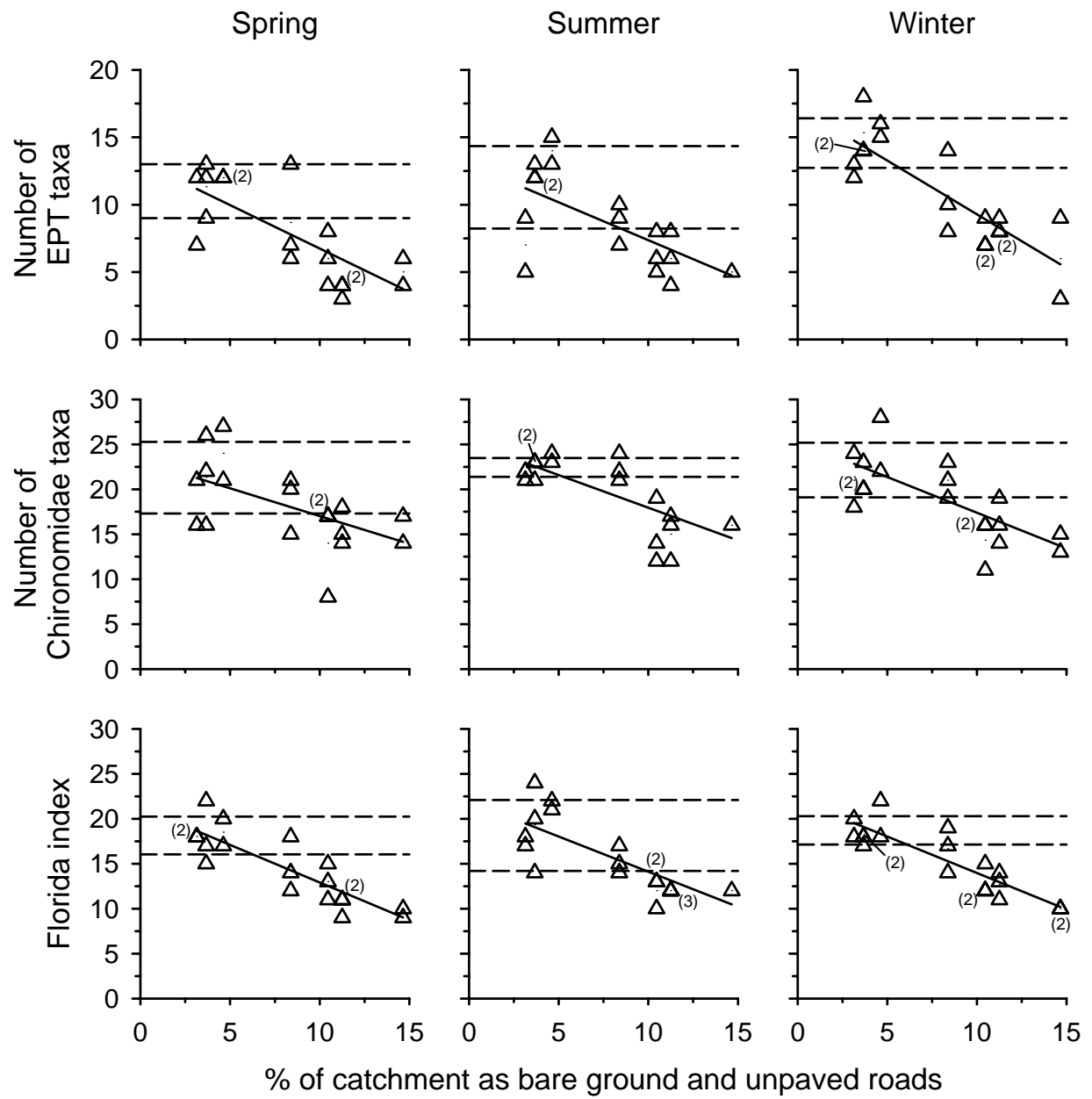


Figure 3.



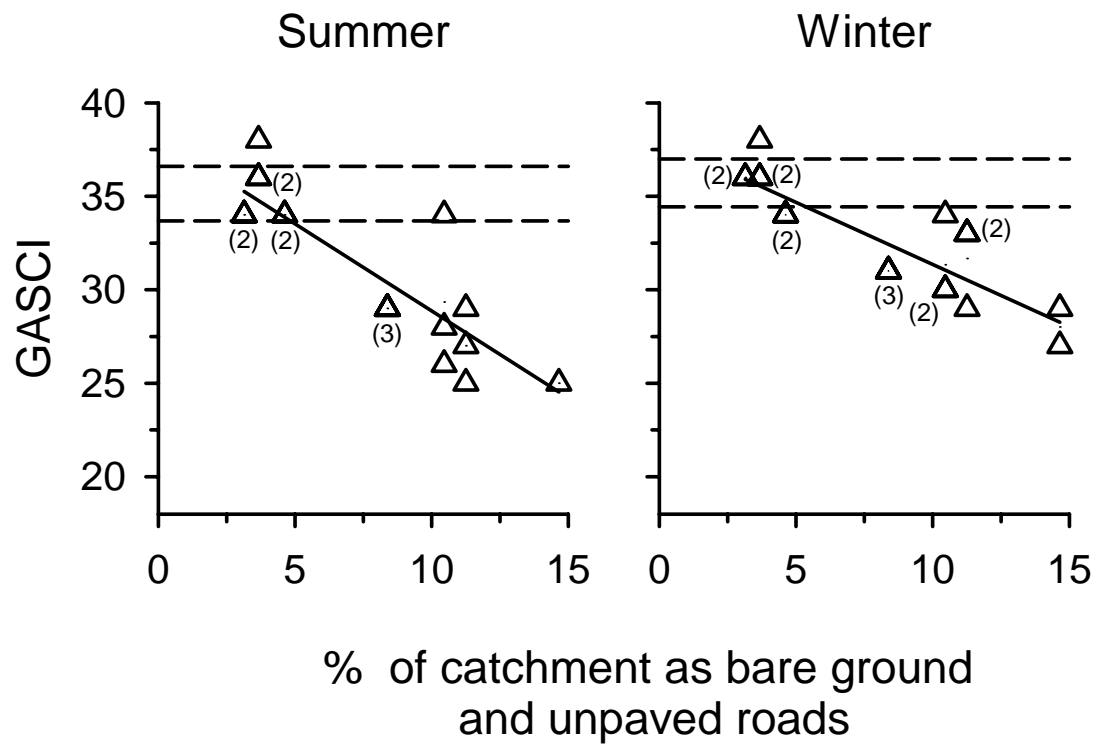


Figure 4.



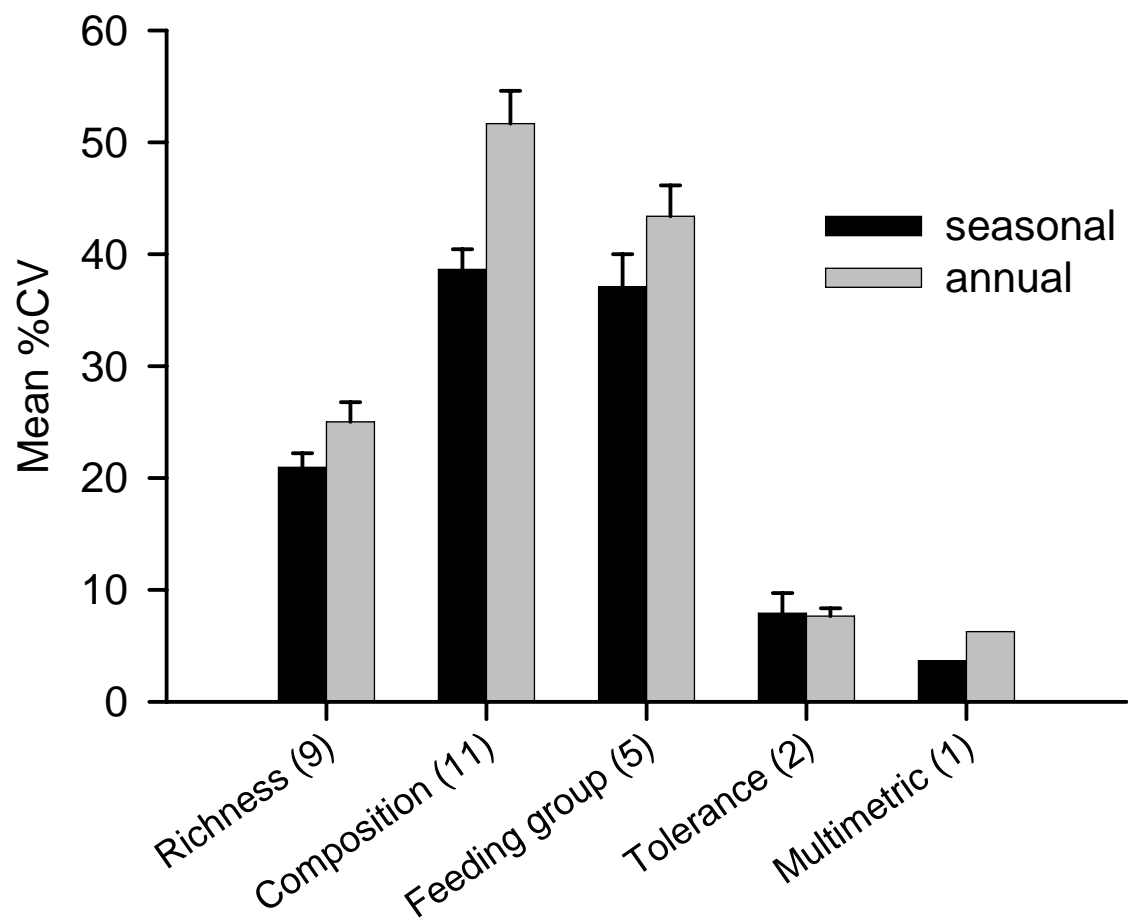


Figure 5.



Appendix. Results of coefficient of variance (%CV) analysis. Abbreviations defined in Table 1.

| Metric                            | within years<br>Coefficient of Variation |       |        | among years<br>Coefficient of Variation |       |        |
|-----------------------------------|--|-------|--------|---|-------|--------|
|                                   | mean                                     | min   | max    | mean                                    | min   | max    |
| No. of Ephemeroptera taxa         | 39.28                                    | 0.00  | 141.42 | 56.99                                   | 0.00  | 173.21 |
| No. of Plecoptera taxa            | 22.03                                    | 0.00  | 34.64  | 26.97                                   | 0.00  | 52.92  |
| No. of Trichoptera taxa           | 38.61                                    | 0.00  | 141.42 | 43.36                                   | 0.00  | 100.00 |
| No. of EPT taxa                   | 22.68                                    | 0.00  | 70.71  | 25.98                                   | 7.69  | 50.94  |
| No. of Chironomidae taxa          | 15.11                                    | 3.01  | 37.12  | 11.69                                   | 4.55  | 23.18  |
| No. of Orthoclaadiinae taxa       | 16.48                                    | 0.00  | 32.73  | 18.79                                   | 9.12  | 30.93  |
| No. of Tanytarsini taxa           | 7.40                                     | 0.00  | 33.33  | 17.80                                   | 0.00  | 43.30  |
| No. of taxa                       | 10.98                                    | 1.30  | 30.88  | 9.47                                    | 1.04  | 17.90  |
| No. of clinger taxa               | 16.02                                    | 4.56  | 38.57  | 14.09                                   | 0.00  | 28.64  |
| Shannon $H'$                      | 6.41                                     | 0.53  | 18.66  | 8.06                                    | 0.84  | 20.97  |
| % Diptera                         | 17.67                                    | 3.28  | 52.43  | 21.18                                   | 2.20  | 42.32  |
| % Ephemeroptera                   | 74.31                                    | 7.41  | 141.42 | 118.09                                  | 41.28 | 173.21 |
| % Plecoptera                      | 48.03                                    | 10.21 | 114.08 | 74.53                                   | 31.67 | 117.85 |
| % Trichoptera                     | 62.34                                    | 1.59  | 141.42 | 88.93                                   | 42.72 | 142.07 |
| % Hydropsychidae of Trichoptera   | 37.89                                    | 2.42  | 141.42 | 59.95                                   | 16.77 | 111.80 |
| % Oligochaeta                     | 55.08                                    | 2.42  | 148.39 | 52.25                                   | 10.76 | 92.34  |
| % Orthoclaadiinae of Chironomidae | 32.06                                    | 3.57  | 84.21  | 33.15                                   | 8.98  | 58.49  |
| % Dominant of total               | 27.64                                    | 1.06  | 57.76  | 27.12                                   | 12.02 | 46.10  |
| % Clingers                        | 30.68                                    | 3.92  | 74.00  | 42.59                                   | 19.28 | 81.05  |
| % Tanytarsini of Chironomidae     | 32.94                                    | 8.56  | 62.81  | 42.66                                   | 11.34 | 103.77 |
| % Predators                       | 26.23                                    | 1.59  | 64.88  | 34.75                                   | 6.75  | 77.53  |
| % Scrapers                        | 58.93                                    | 1.42  | 121.63 | 47.96                                   | 11.10 | 86.09  |
| % Shredders                       | 45.38                                    | 6.56  | 110.70 | 65.53                                   | 20.27 | 110.34 |
| % Filterers                       | 28.14                                    | 6.49  | 90.89  | 37.19                                   | 11.57 | 70.46  |
| % Collector - gatherers           | 26.74                                    | 2.71  | 70.41  | 31.54                                   | 7.72  | 62.93  |
| Florida Index                     | 10.47                                    | 0.00  | 26.03  | 8.64                                    | 0.00  | 18.52  |
| NCBI                              | 5.34                                     | 0.27  | 9.57   | 6.66                                    | 0.42  | 17.75  |
| GASCI                             | 3.66                                     | 0.00  | 14.19  | 6.27                                    | 0.00  | 18.86  |



**Appendix E<sup>7</sup> of Indicators Final Report**

**Influence of catchment disturbance on fish integrity in low-diversity  
headwater streams**

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**Abstract.** An essential step in conserving global freshwater diversity is preservation of headwater streams because they are a dominant form of free-flowing water and are critical determinants of downstream water quality and habitat conditions. We examined relationships between catchment-scale disturbance from military training and stream integrity in headwater streams at the Fort Benning Military Installation, Georgia, USA. We focused on 2 dominant fish populations, Pteronotropis euryzonus (a vulnerable population) and Semotilus thoreauianus (a currently stable population). Proportions of these populations were strongly but oppositely associated with disturbance, with % of the assemblage as P. euryzonus and S. thoreauianus being negatively and positively related to disturbance, respectively. This complementarity likely resulted from different life history traits, feeding behaviors, and habitat preferences between the 2 populations. Additionally, average size of both populations was lower in high-disturbance streams, suggesting both populations were impacted by disturbance. Results also suggest a disturbance threshold, where streams in catchments with disturbance levels ~5–8.1% of the catchment had P. euryzonus proportions below those in low-disturbance streams. Our results suggest that focusing on dominant populations is a viable alternative to ascertain stream integrity and identify potential conservation strategies in low-diversity, headwater streams.



## Introduction

20            Freshwater ecosystems and their communities are among the most imperiled  
systems on earth. The World Wildlife Fund (WWF) estimated a decrease in global  
freshwater biodiversity by ~50% from 1970–2000, a larger decline than that reported for  
terrestrial or marine systems (Loh & Wackernagel 2004). In addition, temperate  
freshwater diversity is declining at a rate faster than tropical diversity (Loh &  
25    Wackernagel 2004). Freshwater vertebrate diversity is dominated by fish, which  
represent >8,500 species (WCMC 1998). Regrettably, there has been a marked  
increase in the number of global extinctions in freshwater fish since 1980 (WCMC  
1998). As a result of the high decline of freshwater diversity, there has been an  
increased awareness for conservation of these systems (Allan & Flecker 1993; Ricciardi  
30    & Rasmussen 1999).

              In the U.S. 39% of streams and rivers are considered impaired (USEPA 2000),  
although 98% of the 5,200,000 km of streams are not of sufficient quality to warrant  
federal protection (Benke 1990). The majority of these stream and river channels  
(~95%) are small headwater streams (e.g., Horton-Strahler 1<sup>st</sup> and 2<sup>nd</sup> order, Leopold et  
35    al. 1964). Often, headwater streams have naturally low-diversity fish assemblages  
(Matthews 1986a, 1998; Grenouillet et al. 2004), which may limit approaches to assess  
impacts from anthropogenic disturbance using traditional community measures such as  
richness and diversity. Because of their high connectivity with associated catchments  
and close proximity to terrestrial sources of disturbance, headwater streams are likely to  
40    be impaired by landscape disturbance to a greater degree than larger streams (Meyer &  
Wallace 2001). Low natural diversity of fish assemblages, and thus limited potential



biotic interactions coupled with potential greater vulnerability makes headwater streams ideal systems to identify mechanisms behind reduced biotic integrity resulting from anthropogenic disturbance.

45           Almost 50% of the ~800 freshwater fish in North America occur in the southeastern U.S. (Lydeard & Mayden 1995; Warren et al. 2000). Many of these species require conservation efforts, as 187 taxa are reported as extinct, endangered, threatened, or vulnerable (Warren et al. 2000). Moreover, since 1989 there has been a 75% increase in jeopardized fish species in the Southeast (Warren et al. 2000), which  
50 suggests stream integrity in this region is decreasing at an alarming rate. The increased number of imperiled species in the Southeast may be a result of the high level of impaired streams in this region (e.g., Alabama = 73%, Georgia = 60%, USEPA, 2000). However, there is a lack of studies addressing conservation strategies for southeastern U.S. fishes, particularly within vulnerable headwater streams. We  
55 examined the degree to which headwater southeastern streams with strongly contrasting catchment disturbance showed correspondingly contrasting fish populations.

## Methods

### Study site

60           We studied seven streams draining catchments at the Fort Benning Military Installation, within the Chattahoochee River Basin, Georgia, USA (Fig. 1). Fort Benning occurs within the Southeastern Plains ecoregion, and contains mainly oak-hickory-pine and southern mixed forests with underlying sandy or sandy clay loam soils (Omernik 1987; Griffith et al. 2001). Land use within study catchments is patchy, consisting of a



mosaic of land used for both military training and forest management, with the main disturbance being bare ground created by mechanized training by tanks and armored personnel carriers, and use of unpaved roads. Study streams were small (1<sup>st</sup>- to 2<sup>nd</sup>- order) and occurred within catchments that varied in the level of military land use and unpaved road use (Maloney et al. in press). Streams channels were narrow (range of wetted width 1.0–2.2 m) and low-gradient (range of channel slope 0.8–2.7%), with abundant sand (range of mean particle size 0.56–0.89 mm, see Maloney et al. in press).

#### Land cover / instream physicochemical variables

We estimated catchment area above sampling locations using a 1993 digital elevation model (DEM, 10-m resolution) and obtained grid coordinates of sampling sites from GPS units. We calculated distance from each site to the mainstem (Upatoi Creek), which we considered a potential colonization source for fish, using a 1999 digitized streams coverage (1:24,000) and summing the distance between each study stream and Upatoi Creek. For our land use component, we defined disturbance as the % of bare ground on slopes >3% summed with the % of unpaved road cover within a catchment (%BGRD), calculated from a 1999 Landsat image (30-m resolution), 1995 road coverage (10 m), and the 1993 DEM (Maloney et al. in press). We quantified all land use / land cover data with Arcview® GIS (Environmental Systems Research Institute, Redlands, California).

Prior to fish sampling at each site, we quantified stream discharge (modified incremental method, see Gore 1996) using a March-McBirney Model 2000 flowmeter. In addition, we quantified average streamwater pH (Beckman model 200 pH meter)



approximately bimonthly from January 2000 to April 2003 and estimated relative abundance of coarse woody debris (CWD) in the stream channel during April 2003. We defined CWD as pieces of wood as well as live roots at least partially submerged >2.5 cm in diameter and used a modified transect method (Wallace & Benke 1984), in which we quantified CWD along 15 transects (spaced 5 m apart) per stream, and expressed abundance as the proportion of streambed covered by CWD.

#### Fish and habitat sampling

We sampled fish assemblages in 3 pool and 3 run microhabitats (riffles were not present) along a 100-m representative reach in each stream in March (spring), July (summer), and December (winter) 2003, using the 2-pass removal-depletion method (Seber 1982) with a backpack electroshocker (Smith-Root LR-24) and block seines. We identified and recorded all fish collected and, except for voucher specimens used for taxonomic confirmation, we returned all individuals to the stream. We also recorded standard length (SL) for all fish collected, and quantified average depth and width (5 subsamples per unit) for each microhabitat type.

Preliminary observations indicated that 2 fish species, broadstripe shiner (Pteronotropis euryzonus) and Dixie chub (Semotilus thoreauianus), were numerically dominant in most of the study streams, therefore we used these species as focal populations to examine relationships between catchment disturbance and assemblage integrity. Both species occur in headwater streams and show broadly overlapping distributions (Metee et al. 1996). However, P. euryzonus is restricted to the Chattahoochee Basin, whereas S. thoreauianus has a broader geographic range, from



the Tombigbee Basin, Alabama, east to the Ochlockonee River, Georgia (Suttkus 1955; Johnston & Ramsey 1990; Boschung & Mayden 2004). Pteronotropis euryzonus is primarily a drift feeder that consumes mostly aquatic insects and detritus (Suttkus 1955), and is typically associated with coarse woody debris in swift, deep water (Suttkus 1955; Boschung & Mayden 2004). Semotilus thoreauianus is a trophic generalist that consumes a variety of aquatic and terrestrial insects, worms, small fish, mollusks, crayfish, and plant detritus, and prefers small, clear streams (Mettee et al. 1996; Boschung & Mayden 2004). Pteronotropis euryzonus broadcasts eggs over vegetation without much parental care (Katula 1993), C. E. Johnston, Auburn University, per. comm.), whereas in S. thoreauianus males excavate nests where females deposit eggs, which are subsequently covered with sediment and then defended by males (Johnston & Ramsey 1990; Maurakis et al. 1993; Boschung & Mayden 2004). Pteronotropis euryzonus is classified as a vulnerable population and S. thoreauianus as currently stable (Warren et al. 2000). Moreover, P. euryzonus is considered rare in Georgia (GADNR 2005).

For these 2 fish populations, we quantified the proportion of the total collected of both species for each stream and season. To satisfy parametric assumptions of normality, we transformed proportional data with arcsine square root and used regression analysis to identify relationships between proportions of each of these populations with catchment disturbance (as %BGRD). In addition, we calculated 95% confidence intervals for the 3 least disturbed streams and used these intervals as a measure of low-disturbance variation, which enabled us to identify disturbance levels where proportions fell outside of the range measured in undisturbed (reference)



streams. We calculated 95% CIs for the 3 least disturbed streams because associated  
 135 catchments all contained  $\leq 5.0\%$  BGRD, which we considered representative of  
 minimally disturbed catchments. In addition, we generated size-frequency distributions  
 for each population in the study streams. However, as a result of low collections in  
 some streams, we pooled specimens collected over the entire study from the 3 lowest  
 and from the 3 highest disturbed catchments, and then tested for differences in size  
 140 frequencies between these high- and low-disturbance categories using a nonparametric  
 analysis of variance by ranks (Kruskal-Wallis test, Zar 1999).

## Results

The proportion of bare ground on slopes  $>3\%$  and unpaved road cover (%BGRD)  
 145 ranged in study catchments from 3.15 to 13.65% (Table 1). Catchment area ranged  
 from 0.72 (Site SB3) to 3.35 (KM1)  $\text{km}^2$ , distance to the Upatoi Creek ranged from 1.73  
 (Site LC) to 26.15 (HB) km and instream CWD relative abundance ranged from 3.3 (Site  
 SB3) to 12.4 (LC) percent of streambed, respectively (Table 1). Average discharge  
 among all study streams was highest in spring ( $0.019 \text{ m}^3/\text{s}$ ), intermediate in winter  
 150 ( $0.014 \text{ m}^3/\text{s}$ ) and lowest in summer ( $0.009 \text{ m}^3/\text{s}$ ), whereas average streamwater pH was  
 consistent among seasons ( $\sim 5.2\text{--}5.4$ , Table 1). Pool volume was highest in winter  
 (mean =  $1.17 \text{ m}^3$ ) intermediate in spring ( $1.02 \text{ m}^3$ ), and lowest in summer ( $0.87 \text{ m}^3$ ),  
 whereas run volume was largest in spring (mean =  $0.85 \text{ m}^3$ ), followed by summer ( $0.76$   
 $\text{m}^3$ ) and winter ( $0.69 \text{ m}^3$ , Table 1).

155 We collected 10 fish species over the study (Table 2). The 2 dominant species,  
Pteronotropis euryzonus and Semotilus thoreauianus, each composed  $>22\%$  of total



fish collected in every season (Table 2), and together they composed >60% of the total fish collected in all 7 streams (Table 3). The remaining 8 species each composed <16% of total fish collected in each season (Table 2). Number of fish collected ranged from 5 to 67, with the fewest collected in BC2 and most in KM1, irrespective of season (Table 3). Season-specific total richness ranged from 1 to 7, with the fewest species collected in SB4 (1 in each season, S. thoreauianus) and the most species in LC (5–7 per season).

Across all seasons, the proportion of the total assemblage as P. euryzonus was strongly negatively related to %BGRD, whereas proportion of the assemblage as S. thoreauianus was strongly positively related to %BGRD (Table 4, Fig. 2). Multivariate models explained additional variation for several seasons. The % of the assemblage as P. euryzonus in spring was negatively related to %BGRD, CWD, and streamwater pH ( $R^2_{\text{adj}} = 0.997$ ), whereas % as P. euryzonus in summer was inversely related to %BGRD and CWD ( $R^2_{\text{adj}} = 0.94$ , Table 4). In contrast, % of the assemblage as S. thoreauianus was positively related to %BGRD and negatively so with average discharge in spring and winter ( $R^2_{\text{adj}} = 0.90$ ,  $R^2_{\text{adj}} = 0.94$ , respectively, Table 4). For spring and summer, at 5.0% BGRD levels the proportion of P. euryzonus fell below the 95% confidence limit for the 3 least-disturbed streams, whereas this threshold occurred at 8.1% BGRD for winter (Fig. 2, top 3 panels). Semotilus thoreauianus showed an opposite pattern, with proportions falling above this measure of low-disturbance conditions at 8.1% BGRD for spring and summer and at 10.5% for winter (Fig. 2, bottom 3 panels).

Average size of both P. euryzonus ( $35.8 \pm 0.73$  mm, mean  $\pm$  SE) and S. thoreauianus ( $49.3 \pm 1.43$  mm) was significantly different between streams in high- vs.



low-disturbance catchments (Fig. 3). Standard length of P. euryzonus was lower in high-disturbance streams ( $26.0 \pm 1.35$  mm) than low-disturbance streams ( $SL=37.9 \pm 0.87$  mm;  $\chi^2_1 = 26.50$ ,  $p < 0.0001$ ), and SL of S. thoreauianus followed the same trend (i.e.,  $SL=42.9 \pm 1.32$  mm vs.  $77.7 \pm 4.9$  mm in high- vs. low-disturbance streams, respectively;  $\chi^2_1 = 35.18$ ,  $p < 0.0001$ ).

## Discussion

Landscape alteration within catchments can cause severe changes in stream physicochemical conditions, which in turn can alter stream community structure, including fish assemblages. Catchment land use is often among the best predictors of fish assemblage integrity, and multimetric measures of relative abundance and functional group composition such as the Index of Biotic Integrity (IBI) often signal impairment from landscape disturbance (Karr 1991; Allan et al. 1997). Unfortunately, the low number of species we collected (10, mean ~3 species / stream) and high relative abundance of the 2 numerically dominant populations, P. euryzonus and S. thoreauianus, precluded our use of such measures, as IBIs are highly sensitive to low richness and high dominance (Osborne et al. 1992; Schleiger 2000). The limitation of this approach is generally true for many low-diversity headwater streams. We suggest that population-based measures provide more accurate measures of stream integrity in these naturally low-diverse streams than assemblage based measures. In our study, we also experienced a potential disturbance threshold at ~5–8.1% of the catchment as bare ground on slopes >3% and unpaved roads. One potential conservation strategy in streams at Fort Benning, which may be applicable to other low-gradient Southeast



streams, is to limit the amount of catchment disturbance to levels that remain below apparent thresholds.

We hypothesize that the fundamentally different habitat requirements and life history traits account for the diametrically opposite distribution patterns of the 2 focal fish populations with respect to catchment disturbance. For example, streambed stability and coarse woody debris decreases with increasing catchment disturbance in these streams (Maloney et al. in press), a relationship which likely provided fewer spawning sites and preferred habitat for P. euryzonus. Reductions in CWD abundance in stream channels may decrease pool depth (Angermeier & Karr 1984), which may result in reduced habitat quantity and quality for P. euryzonus. In fact, over the entire study mean % P. euryzonus of total individuals collected was positively related to CWD ( $R^2 = 0.68$ ,  $P = 0.02$ ,  $\beta = 8.24$ ). A potential conservation strategy would be to increase instream CWD to levels comparable to amounts measured in streams within low disturbance catchments, as this would likely increase bed stability and provide essential habitat for P. euryzonus.

Dissimilar food and habitat requirements may account for lower abundances of P. euryzonus and the higher abundances of S. thoreauianus in high-disturbance streams, but they do not account for low relative abundance of S. thoreauianus in low-disturbance streams. Juvenile S. thoreauianus (20–40 mm SL) may consume similar prey as adult P. euryzonus and both prefer similar habitats (Barber & Minckley 1971; Ross et al. 2001), so it is possible that juvenile S. thoreauianus are competitively inferior to adult P. euryzonus, and are displaced from sections where P. euryzonus occur (i.e., low disturbed streams). In fact, at sites where the 2 populations were sympatric



individuals of S. thoreauianus were larger (mean = 55.9 mm SL) than those individuals collected from sites without P. euryzonus (mean = 45.0 mm SL,  $\chi^2_1 = 11.16$ ,  $p = 0.0008$ ). If this pattern is real and P. euryzonus is sensitive to sedimentation and does not survive in highly disturbed streams, then high abundance of S. thoreauianus in disturbed streams may represent a competitive release. The role of disturbance in sustaining competitively inferior or non-indigenous populations in the presence of superior competitors has been reported (McAuliffe 1984; Resh et al. 1988; Byers 2002). However, interspecific interactions between P. euryzonus and S. thoreauianus have not been observed so the degree to which catchment disturbance alters important resources and conditions for these species and thus modify competitive dominance is unknown.

Dramatic differences in size distributions between the most- and least-disturbed streams for both species, with significantly smaller individuals in the most-disturbed streams, provide strong circumstantial evidence that both populations were negatively affected by disturbance from military land use. At least 2 explanations exist for this pattern. First, size distribution data implied that both P. euryzonus and S. thoreauianus were capable of colonizing and/or recruiting in disturbed streams although, relative to low-disturbance sites, both species exhibited low survivorship in, or high adult emigration from, high-disturbance sites. Either or both mechanisms could result from decreased habitat (cover, spawning habitat) and/or food availability attributable to increased sediment disturbance (Ritchie 1972; Angermeier & Karr 1984; Ryan 1991; Sutherland et al. 2002), which over time could decrease prevalence of adults. Decreased habitat availability appears particularly likely in this regard, as available



habitat in terms of pool size and abundance of CWD (resting habitat, potential refugia  
 250 from predation) decreased with increasing catchment disturbance. Second, in our sites  
 stream flashiness (i.e., rapid fluctuations in stream stage in response to storms)  
 increased with catchment disturbance (Maloney et al. in press), which in disturbed  
 streams may have transported larger and potentially more vulnerable fish downstream,  
 or increased their mortality, thus skewing populations to smaller sizes. Rapid increases  
 255 in stream stage, perhaps separately or in combination with decreased cover, may  
 increase fish mortality and/or emigration to downstream reaches (Matthews 1986b;  
 Yount & Niemi 1990; Grossman et al. 1998).

The preservation and restoration of headwater streams is an essential first step  
 in the conservation of global freshwater diversity because these streams 1) are the  
 260 dominant form of free-flowing water in terrestrial landscapes, and 2) are critical  
 determinants of water quality and habitat conditions in larger downstream systems  
 (Meyer and Wallace 2001). As such, conservation efforts of larger systems likely will be  
 more effective if upland stressors (e.g., sediment, nutrients, etc.) are properly managed  
 closer to the source. Our findings suggest 2 potential strategies to conserve stream  
 265 integrity in low-diversity headwater streams that may be applicable elsewhere: 1) limit  
 catchment-scale disturbance to levels below identified thresholds, and 2) restoration of  
 limiting resources (i.e., habitat) necessary for population sustainability.



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395 Table 1. Summary of land use, reach and habitat scale variables. %BGRD = the percentage of bare ground on slopes >3% and road cover within a study catchment, CWD = coarse woody debris relative abundance. Data for CWD, pH, pool volume, and run volume are season mean (SE).

| Stream                 | Abbreviation | %BGRD | Length to                            |                      | CWD<br>(% stream<br>bottom coverage) | Season | Discharge<br>(m <sup>3</sup> /sec) | pH          | Pool volume<br>(m <sup>3</sup> ) | Run volume<br>(m <sup>3</sup> ) |
|------------------------|--------------|-------|--------------------------------------|----------------------|--------------------------------------|--------|------------------------------------|-------------|----------------------------------|---------------------------------|
|                        |              |       | Catchment<br>area (km <sup>2</sup> ) | Upatoi<br>Creek (km) |                                      |        |                                    |             |                                  |                                 |
| Bonham Tributary       | BC2          | 3.15  | 0.75                                 | 2.05                 | 10.1 (2.2)                           | Spring | 0.005                              | 4.89 (0.05) | 0.65 (0.07)                      | 0.30 (0.08)                     |
|                        |              |       |                                      |                      |                                      | Summer | 0.001                              | 4.92 (0.03) | 0.50 (0.13)                      | 0.55 (0.31)                     |
|                        |              |       |                                      |                      |                                      | Winter | 0.005                              | 5.24 (0.19) | 0.67 (0.04)                      | 0.31 (0.07)                     |
| Sally Branch Tributary | SB2          | 8.12  | 1.23                                 | 11.92                | 8.7 (1.6)                            | Spring | 0.027                              | 5.92 (0.13) | 0.62 (0.10)                      | 0.40 (0.13)                     |
|                        |              |       |                                      |                      |                                      | Summer | 0.009                              | 6.24 (0.03) | 0.38 (0.02)                      | 0.61 (0.05)                     |
|                        |              |       |                                      |                      |                                      | Winter | 0.016                              | 6.03 (0.06) | 0.86 (0.39)                      | 0.29 (0.02)                     |
| Sally Branch Tributary | SB3          | 10.49 | 0.72                                 | 12.80                | 3.3 (0.9)                            | Spring | 0.007                              | 6.06 (0.13) | 0.28 (0.06)                      | 0.14 (0.02)                     |
|                        |              |       |                                      |                      |                                      | Summer | 0.004                              | 6.52 (0.06) | 0.20 (0.01)                      | 0.22 (0.08)                     |
|                        |              |       |                                      |                      |                                      | Winter | 0.008                              | 6.13 (0.06) | 0.34 (0.02)                      | 0.31 (0.04)                     |
| Sally Branch Tributary | SB4          | 13.65 | 1.00                                 | 12.86                | 3.6 (1.5)                            | Spring | 0.012                              | 5.45 (0.19) | 0.32 (0.07)                      | 0.31 (0.04)                     |
|                        |              |       |                                      |                      |                                      | Summer | 0.006                              | 5.78 (0.15) | 0.35 (0.06)                      | 0.24 (0.03)                     |



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|                               |     |      |      |       |            |        |       |             |             |             |
|-------------------------------|-----|------|------|-------|------------|--------|-------|-------------|-------------|-------------|
| Hollis Branch                 | HB  | 6.62 | 2.15 | 26.15 | 6.5 (2.2)  | Winter | 0.009 | 5.54 (0.19) | 0.38 (0.08) | 0.29 (0.04) |
|                               |     |      |      |       |            | Spring | 0.018 | 5.13 (0.08) | 1.65 (0.46) | 0.76 (0.10) |
|                               |     |      |      |       |            | Summer | 0.013 | 5.43 (0.04) | 0.74 (0.06) | 0.93 (0.21) |
| Kings Mill Creek<br>Tributary | KM1 | 5.01 | 3.35 | 3.11  | 7.5 (1.1)  | Winter | 0.018 | 5.08 (0.04) | 2.24 (0.83) | 1.02 (0.09) |
|                               |     |      |      |       |            | Spring | 0.037 | 4.95 (0.06) | 1.35 (0.34) | 1.12 (0.17) |
|                               |     |      |      |       |            | Summer | 0.020 | 5.07 (0.03) | 2.14 (0.58) | 1.56 (0.22) |
| Lois Creek                    | LC  | 3.67 | 3.32 | 1.73  | 12.4 (2.1) | Winter | 0.029 | 4.98 (0.05) | 1.94 (0.17) | 0.98 (0.25) |
|                               |     |      |      |       |            | Spring | 0.044 | 4.87 (0.02) | 2.49 (0.61) | 3.47 (0.95) |
|                               |     |      |      |       |            | Summer | 0.013 | 4.89 (0.07) | 1.97 (0.21) | 1.43 (0.46) |
|                               |     |      |      |       |            | Winter | 0.022 | 5.04 (0.07) | 1.68 (0.19) | 1.85 (0.69) |

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Table 2. Absolute and relative abundance of fish species collected during the study.

400

| Family          | Species                        | Common name            | Number collected |        |        | % of total collected |        |        |
|-----------------|--------------------------------|------------------------|------------------|--------|--------|----------------------|--------|--------|
|                 |                                |                        | Spring           | Summer | Winter | Spring               | Summer | Winter |
| Aphredoderidae  | <u>Aphredoderus sayanus</u>    | Pirate Perch           | 8                | 3      | 5      | 2.6                  | 1.7    | 2.0    |
| Centrarchidae   | <u>Lepomis gulosus</u>         | Warmouth               | 0                | 0      | 1      | 0.0                  | 0.0    | 0.4    |
|                 | <u>Lepomis miniatus</u>        | Redspotted Sunfish     | 3                | 1      | 1      | 1.0                  | 0.6    | 0.4    |
| Cyprinidae      | <u>Notemigonus crysoleucas</u> | Golden Shiner          | 0                | 1      | 1      | 0.0                  | 0.6    | 0.4    |
|                 | <u>Pteronotropis euryzonus</u> | Broadstripe Shiner     | 67               | 69     | 117    | 22.0                 | 39.2   | 46.1   |
|                 | <u>Semotilus thoreauianus</u>  | Dixie Chub             | 90               | 67     | 81     | 29.5                 | 38.1   | 31.9   |
| Esocidae        | <u>Esox americanus</u>         | Grass Pickerel         | 1                | 5      | 2      | 0.3                  | 2.8    | 0.8    |
| Ictaluridae     | <u>Ameiurus natalis</u>        | Yellow Bullhead        | 0                | 4      | 2      | 0.0                  | 2.3    | 0.8    |
| Percidae        | <u>Percina nigrofasciata</u>   | Blackbanded Darter     | 1                | 1      | 2      | 0.3                  | 0.6    | 0.8    |
| Petromyzontidae | <u>Ichthyomyzon gagei</u>      | Southern Brook Lamprey | 48               | 14     | 22     | 15.7                 | 8.0    | 8.7    |
| Total:          |                                |                        | 218              | 165    | 234    |                      |        |        |



Table 3. Number of fish collected and species richness by stream and season and proportion of total fish collected as P. euryzonus and S. thoreauianus. Stream abbreviations defined in Table 1.

| Stream       | Number collected |        |        | Total taxa |        |        | % as <u>P. euryzonus</u>   |
|--------------|------------------|--------|--------|------------|--------|--------|----------------------------|
| abbreviation | Spring           | Summer | Winter | Spring     | Summer | Winter | and <u>S. thoreauianus</u> |
| BC2          | 14               | 5      | 6      | 3          | 2      | 2      | 96.0                       |
| SB2          | 16               | 32     | 35     | 2          | 5      | 4      | 85.5                       |
| SB3          | 17               | 18     | 34     | 1          | 4      | 4      | 94.2                       |
| SB4          | 12               | 22     | 23     | 1          | 1      | 1      | 100.0                      |
| HB           | 67               | 22     | 35     | 5          | 6      | 5      | 79.8                       |
| KM1          | 67               | 41     | 44     | 3          | 4      | 6      | 62.5                       |
| LC           | 25               | 25     | 57     | 7          | 7      | 5      | 74.8                       |



Table 4. Selected models of multiple regression analysis on the abundance of  
 405 Pteronotropis euryzonus and Semotilus thoreauianus relative to the total fish collected.

Parameter abbreviations: CWD = relative abundance of coarse woody debris, Q = discharge, %BGRD = % of bare ground on slopes > 3% and unpaved road cover within catchment, L\_Upatoi = linear distance to the Upatoi Creek mainstem.

|                                   |   | Adjusted          |         |
|-----------------------------------|---|-------------------|---------|
| Variable                          | Parameters in model                             | $\underline{R}^2$ | p value |
| % <u>P. euryzonus</u> of total    |   |                   |         |
| Spring                            | pH (− 0.71)                                     | 0.83              | 0.003   |
|                                   | %BGRD (− 0.09)                                  | 0.76              | 0.007   |
|                                   | %BGRD (− 0.05), pH (− 0.45)                     | 0.96              | 0.0009  |
|                                   | %BGRD (−0.08), pH (− 0.46), CWD (− 3.76)        | 1.00              | 0.0003  |
| Summer                            | %BGRD (− 0.10)                                  | 0.84              | 0.003   |
|                                   | %BGRD (− 0.15), CWD (− 7.24)                    | 0.94              | 0.002   |
|                                   | %BGRD (− 0.15), CWD (− 8.93), L_Upatoi (− 0.01) | 0.98              | 0.001   |
| Winter                            | %BGRD (− 0.10)                                  | 0.86              | 0.002   |
|                                   | CWD (11.26)                                     | 0.79              | 0.005   |
|                                   | %BGRD (− 0.07), CWD (4.71)                      | 0.88              | 0.007   |
|                                   | %BGRD (− 0.07), CWD (6.98), L_Upatoi (0.01)     | 0.96              | 0.007   |
| % <u>S. thoreauianus</u> of total |   |                   |         |
| Spring                            | %BGRD (0.13)                                    | 0.73              | 0.009   |
|                                   | %BGRD (0.10), Q (− 16.47)                       | 0.90              | 0.004   |
|                                   | %BGRD (0.07), Q (− 15.49), pH (0.36)            | 0.96              | 0.004   |
| Summer                            | %BGRD (0.11)                                    | 0.83              | 0.003   |
|                                   | %BGRD (0.17), CWD (7.65)                        | 0.90              | 0.005   |



|        |                           |      |       |
|--------|---------------------------|------|-------|
|        | %BGRD (0.11), Q (− 16.50) | 0.86 | 0.009 |
| Winter | %BGRD (0.12)              | 0.80 | 0.004 |
|        | %BGRD (0.10), Q (− 24.46) | 0.94 | 0.001 |

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## Figure Legends

Figure 1. Locations of study catchments (polygons) within Fort Benning Military Installation, GA. Dotted line in middle figure represents the Chattahoochee River, which separates Alabama (AL) and Georgia (GA) (Modified from Maloney et al. submitted). Numbers in the right figure identify watersheds on the same stream (e.g., 2 and 3 on the Sally Branch represent Sally Branch Tributaries 2 and 3, SB2 and SB3, respectively).

Figure 2. Relationship between the proportion of Pteronotropis euryzonus (top 3 panels) and Semotilus thoreauianus (bottom 3 panels) of total individuals collected with the proportion of bare ground on slopes >3% and unpaved road cover for spring, summer, and winter 2003. Curved lines are 95% confidence intervals. Solid lines represent trends using means, dashed lines represent 95% upper and lower confidence limits of the 3 least disturbed streams in the data set (BC2, KM1, LC). See Table 4 for  $R^2_{adj}$  and p-values.

Figure 3. Size class frequency distributions of Semotilus thoreauianus (left 2 panels) and Pteronotropis euryzonus (right 2 panels) in the 3 least-disturbed (top panel) and 3 most-disturbed (bottom panel) study streams.



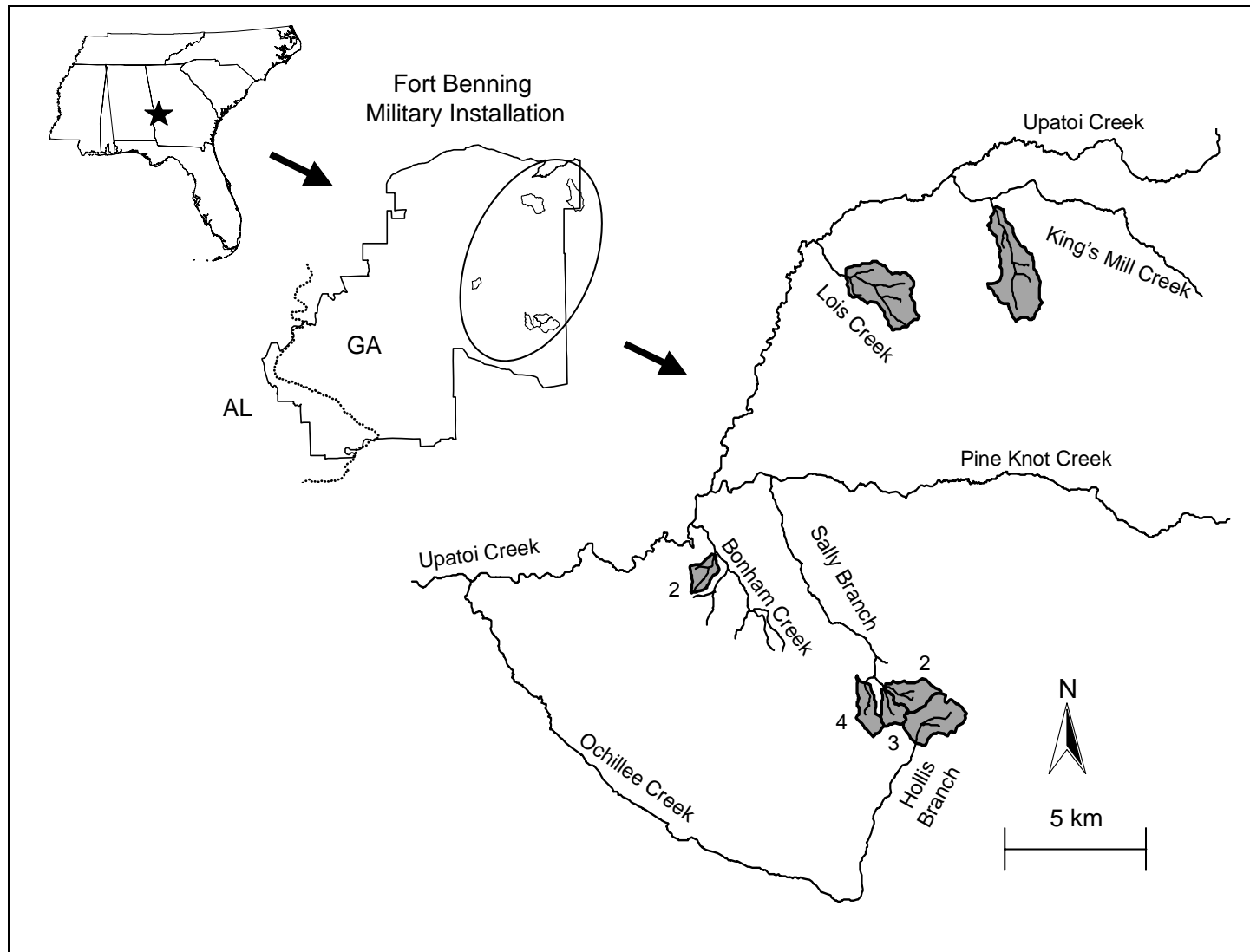


Figure 1



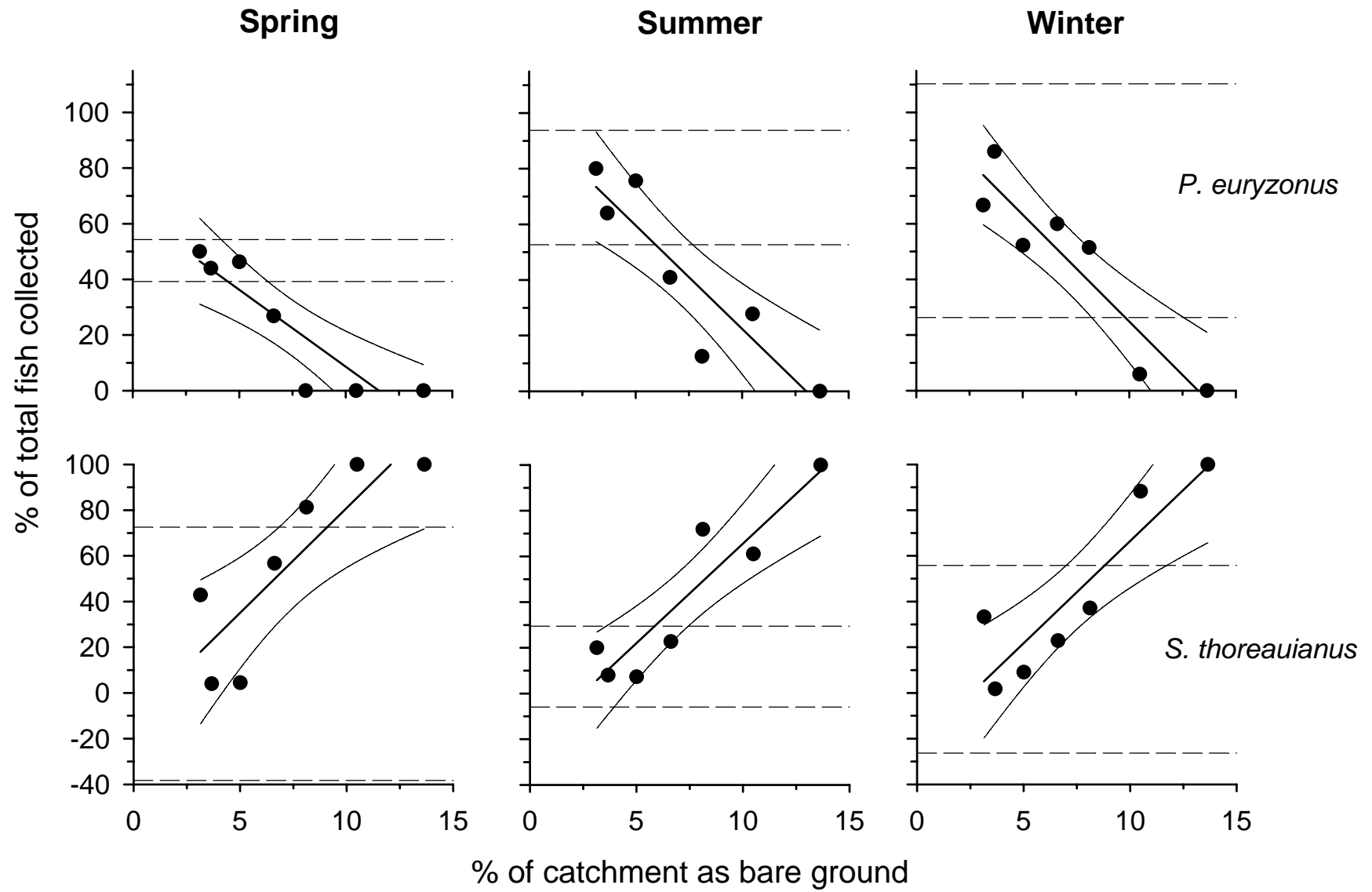


Figure 2.



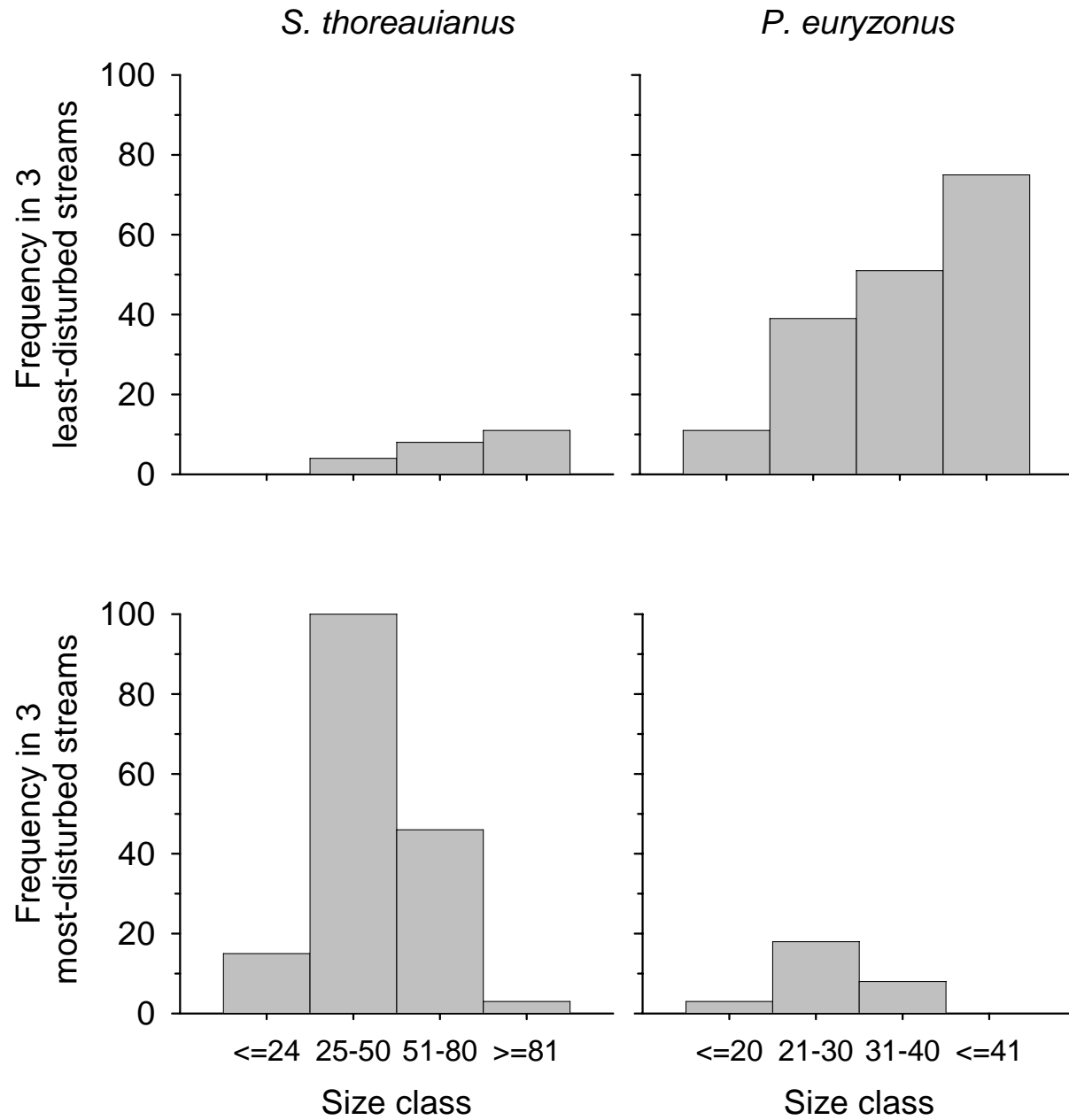


Figure 3



Land use legacies and small streams: the relationships between historic and contemporary land use patterns with present-day stream conditions

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## **Abstract**

Although contemporary stream conditions are products of both historic and present-day catchment conditions, most stream studies focus only on contemporary catchment conditions. We present results from a multi-year, multi-disciplinary research project aimed at identifying the relationships between contemporary and historic land use with present-day stream conditions. We sampled twelve 2<sup>nd</sup>-and 3<sup>rd</sup>-order streams on the Fort Benning Military Installation, Georgia, USA from 2000 to 2003. Prior to military purchase (pre-1942) these catchments were highly influenced by intensive agriculture and silviculture; post-military purchase, catchments have been used for military training and forest management. Contemporary streamwater chemistry (pH, soluble reactive phosphorus), organic matter variables (benthic particulate organic matter, coarse woody debris, dissolved organic carbon), and stream flashiness were more related to contemporary than historic land use patterns, whereas streambed instability was strongly related to historic land use. Diatom measures (% acidobiontic and % motile diatoms) were equally related to both contemporary and historic disturbance, suggesting diatoms were influenced by both historic and contemporary land use. Macroinvertebrate measures in which chironomids were highly influential (total taxa, diversity, number chironomid taxa) were best explained by contemporary land use. Macroinvertebrate measures incorporating larger and longer-lived taxa (EPT, % clingers, density) and fish assemblage measures (total taxa, diversity, total number collected) all were related to historic land use, collectively indicating that larger and longer-lived taxa were more susceptible to historic land use. Whole-stream respiration was related to contemporary land use, whereas gross primary production was related to



historic land use and catchment size. Taken together, these results suggest most stream physicochemical variables recover from historic disturbance within a 60 y time period. However, bed stability is highly influenced by historic terrestrial disturbance and may take longer than 60 y to restabilize. Further, results suggest susceptibility of biota to historic land use may depend on body size and turnover rate, with larger and low turnover taxa being more susceptible to legacy effects.

Keywords: land use legacy, streams, water chemistry, physicochemistry, coarse woody debris (CWD), particulate organic matter (POM), macroinvertebrates, fish



## Introduction

The role of land use legacy effects on contemporary ecosystem structure and function has been receiving increasing attention, likely as a result of the important influence of historic disturbance on contemporary conditions (Foster et al. 2003). There is ample evidence illustrating the influence of historic land use on contemporary forest ecosystems, especially concerning vegetation patterns. In general, compared to historically undisturbed sites, forest canopies in historically disturbed plots are more homogenized having a higher proportion of shade-intolerant, faster growing species (Zimmerman et al. 1995, Foster et al. 2003, Grau et al. 2003, Ziegler 2004). Moreover, seed banks within these historically disturbed forests have a higher abundance of early successional seeds than non-disturbed forests (Bossuyt and Hermy 2001). The imprint of historic disturbance is also reflected in forest soils. For example, soils within historically disturbed agricultural lands have lower organic matter, nitrogen, carbon and pH levels than historically undisturbed areas (Burke et al. 1995, Verheyen et al. 1999, Richter et al. 2000). These differences are hypothesized to endure for decades to centuries following cessation of land use (Burke et al. 1995, Compton et al. 1998, Knops and Tilman 2000). The soil microbial community is also susceptible to historic land use exhibiting lower biomass and altered community structure decades after abandonment (Burke et al. 1995, Steenwerth et al. 2003). As streams are highly dependent on surrounding catchment conditions, legacy effects to terrestrial ecosystems likely impact associated streams.

As human alterations to catchment conditions may last for centuries to millennia (Koerner et al. 1997, Dupouey et al. 2002), they indirectly influence stream conditions



for years post disturbance. For example, gully formation and incision of channels often occur as a result of poor agriculture and urbanization practices (Ireland et al. 1939, Paul and Meyer 2001), which may take centuries or longer for stream recovery. Further, channel coarse woody debris (CWD), which stabilizes channels and serves as habitat for stream biota (Smock et al. 1989, Wallace et al. 1995), is the result of historic inputs from surrounding vegetation that may be decades to centuries old (Hyatt and Naiman 2001, Wallace et al. 2001). Moreover, although streamwater chemistry has widely been reported to return to pre-disturbed levels within 10–15 y (see Likens et al. 1978, Lynch and Corbert 1991), Swank et al. (2001) report elevated  $\text{Ca}^+$  and  $\text{NO}_3^-$  ~20 y following forest harvest. In a study designed to examine the relationship between historic land use and contemporary stream physicochemistry Scott et al. (2002) report that stream chemistry is related more to contemporary whereas sediment more related to historic land use patterns.

The legacy of land use likely affects stream biota as well. However little research has directly addressed this issue, most likely a result of limited long-term biological and land use data. In one of the few studies, Harding et al. (1998) reported that catchment land use in the 1950s was a better predictor of both contemporary macroinvertebrate and fish diversity than was 1990s land use. Sutherland et al. (2002), however, reported no such relationship between contemporary fish assemblage structure and historic land use a discrepancy attributed to different historical land use intensities. There is however, abundant evidence supporting legacy effects on fish assemblages as fish have been widely reported to have been negatively impacted by past human actions (Pflieger and Grace 1987, Rahel 2000). Unfortunately, there is a general lack of



evidence supporting legacy effects for other trophic levels (e.g., periphyton, microbes, macroinvertebrates). However, all of the above assemblages are affected by recent human perturbation (Resh et al. 1988, Paul and Meyer 2001, Iwata et al. 2003), thus the existence of legacy assemblages at these trophic levels is highly probable. A assessment of historic land use effects on multi-trophic levels is needed because extrapolations from higher trophic levels to lower ones is not applicable due to the high variation of life history traits among different trophic levels.

The concept of land use legacies has vast implications for cross-stream contemporary comparisons because present day differences may actually be a result of historic rather than current or recent disturbance. For example, studies that incorporate the influence of current catchment-scale disturbance on contemporary biotic assemblages may misidentify potential stressors and relationships because of the presence of a disturbance legacy effect. Such misidentification also may apply to ecological studies investigating local-scale conditions, as a “legacy assemblage” would likely result in a misidentification of possible ecological responses of biota to their surrounding environment. Legacy effects also may affect sensitivity of measures of biotic integrity, as they may lead to misclassification of contemporary catchment disturbance classes (e.g., a historically disturbed [depauperate] stream showing catchment, but not biotic, recovery). However, most studies report only relationships between contemporary land use and stream conditions, thus failing to identify the possible effects of land use legacies. The objective of this study was to report whether contemporary or historic land use patterns were better predictors of present-day stream physicochemical and biological conditions. Specifically, we related present-day stream



habitat structure, water chemistry, whole-stream respiration and gross primary production, and diatom, macroinvertebrate, and fish assemblages to historic (~60 y) and contemporary land use patterns.

## Study site

Our study was conducted on the Fort Benning Military Installation (FBMI), which is located in the Southeastern Plains ecoregion of Chattahoochee and Muscogee Counties, of Georgia, USA (Fig. 1). Fort Benning comprises approximately 735 km<sup>2</sup> with a local climate that is humid and mild with precipitation occurring throughout the year (average = 105 cm). The majority of the base is dominated by pine (i.e., longleaf (*Pinus palustris*), short leaf (*Pinus echinata*) and loblolly (*Pinus taeda*)) and mixed hardwood (dominated by oak) communities. Streams are of low gradient, mainly sandy bottomed, with large amounts of woody debris and organic matter (i.e. leaf detritus), with many meanders and broad floodplains dominated by black gum (*Nyssa sylvatica*).

The US military purchased the portions of the base studied here by 1942. Prior to military purchase, land use practices were primarily agricultural and silvicultural, which persisted from the early 1800s up to military purchase (Hilliard 1984, Frost 1993, Kane and Keeton 1998). Since military purchase, the land has been used for infantry and mechanized training with associated heavy equipment vehicles (including tanks, armored personnel carriers, and a variety of light and heavy wheeled vehicles). Both training practices dramatically alter the landscape by disrupting the surface soil layer and vegetative cover (USAIC 2001, Dale et al. 2002). However, large areas of these



lands are also used solely for foot-traffic training and thus allowed to revegetate and serve as contemporary reference areas (Dale et al. 2002). Established compartments at FBMI segregate these military training activities, thus it was possible to select sites that varied in the level of contemporary disturbance from military training and under different levels of catchment revegetation from historic agricultural disturbance. Further, controlled burning and selective timber harvesting are frequently used and well documented on the base.

We studied twelve small 2<sup>nd</sup> and 3<sup>rd</sup> order catchments (area ranged from 0.33 to 5.43 km<sup>2</sup>) on the eastern part of FBMI (Fig. 1, Table 1). Our study sites were typical sandy Southeastern Plains streams (see Maloney et al. In press). Our contemporary low-disturbed sites were mostly within compartments reserved for light infantry training, whereas the majority of contemporary heavy-disturbed sites were within compartments reserved for heavy-tracked vehicle training.

## **Methods**

Our multidisciplinary approach enabled sampling of a suite of traditionally measured stream response variables to catchment disturbance. Physicochemical variables measured were stream hydrologic flashiness, bed-instability, bed organic matter content and coarse woody debris. We also measured streamwater NO<sub>3</sub>-N, soluble reactive phosphorus (SRP), dissolved organic carbon (DOC), and pH. Biotic components consisted of whole-stream community respiration and gross primary productivity (GPP) as well as stream diatom, macroinvertebrate, and fish assemblages. Due to the vast array of contemporary stream measures, sampling of each measure



occurred at different intervals and years (Table 1). For example, we sampled macroinvertebrates seasonally from 2000 to 2002 and fish seasonally in 2003 (Table 1).

### Land Use Classification

We quantified land use within study catchments using ArcView© 3.2 software (Environmental Systems Research Institute, Redlands, California) and the ArcView© extension Analytical Tools Interface for Landscape Assessments (ATtILA, Ebert and Wade 2000). Land use during 1999 was derived from satellite imagery (Landsat 28.5 m, July and December 1999), whereas 1944 land use was derived from aerial photography (1 m, March 1944). Digital elevation models (DEMs, 1:24,000, grid size 10 m, 1993) were used in conjunction with each land use coverage to calculate an estimate of disturbed land (%**D**, as the % of disturbed land (bare ground, fields) on slopes > 3% and under unpaved road cover) as well as the proportion as recovering land (e.g., shrubs, sparse vegetation) on slopes > 3% (%**R**) in a catchment for each time period as well as estimate drainage area above our sampling sites (Maloney et al. In press). The 1944 land use conditions reflected agricultural activities at the time this portion of the base was purchased by the military, therefore **D44** represented recently abandoned agricultural lands, whereas **R44** included lands recovering from fields abandoned prior to 1944 (Figure 2). The 1999 land use represents land under military ownership for ~60 years, thus land use included in **99D** was reflective of heavy machinery and munitions military training, whereas **99R** included land recovering from historic agricultural and silvicultural practices as well as historic military training maneuvers (Figure 2).



## Physicochemical variables

Streambed instability, flashiness, and organic matter. —

We estimated streambed instability by quantifying sediment movement using 5 leveled cross-stream transects per stream. Each transect was constructed of 2 pieces of metal on opposite stream banks. During initial deployment metal bars were leveled and marked using plastic cable ties. Measurements of distance between the horizontal transect and the stream bottom were then taken in January 2003 and July 2003 at fixed 20 cm intervals along each leveled transect. We estimated streambed instability as the average absolute change in these distances between the 2 time periods (see Maloney et al. In press).

Stream flashiness, defined here as hydrologic response to a precipitation event, was estimated by calculating recession coefficients of the receding limb of storm hydrographs (Domenico and Schwartz 1990, Rose and Peters 2001). Discharge for storm events was measured using an ISCO ultrasonic flow module (model 750) and series portable sampler (model 6700). Recession coefficients were calculated as the slope of the regression of the natural logarithm of discharge measured at ½ to 1 h intervals over the first 4 h following peak discharge.

Streambed benthic particulate organic matter (BPOM) was sampled in the thalweg at 3 sites per stream every 2 months from August 2001 to May 2003 using replicate sediment cores (2.01 cm<sup>2</sup> × 10 cm long PVC tubing). We estimated BPOM as ash-free dry mass (AFDM, the difference between dried (80° C, 2-3 days) and ashed (550 °C, 3 h) weights). Coarse woody debris (CWD) was quantified in April 2002 and



March 2003 using a modified transect method along 15 transects (spaced 5 m apart) in each stream. We quantified all submerged CWD >2.5 cm in diameter within a 1-m wide band at each cross-stream transect and converted to an areal estimate (see Maloney et al. In press).

We measured non-storm flow stream water  $\text{NO}_3\text{-N}$ , SRP, and DOC concentrations ~6 times each year from September 2000 to September 2003 using grab water samples (1 L Nalgene bottles). Samples were shipped on ice to the Oak Ridge National Laboratory, Oak Ridge, TN, and analyzed for SRP (spectrophotometer, molybdate blue method and a 10 cm light path),  $\text{NO}_3\text{-N}$  (colormetrically using a Bran Lubbe autoanalyzer), and DOC (high temperature combustion using a Shimadzu Model 5000 TOC analyzer after acidification and purging to remove inorganic carbon). We measured pH (Beckman model 200 pH meter) approximately bimonthly from January 2000 to April 2003 (Houser, unpublished data).

### Stream biota

Diatoms. — We sampled diatoms in 11 streams 2 × per y (spring, summer) from September 2001 to May 2002 at 3 sites per stream. Three run sections per site were sampled by inverting a 4.7 cm-diameter petri dish over the streambed and removing trapped sand and sediments (Barbour et al 1999). Samples from each site were homogenized, preserved in 2% gluteraldehyde, cleaned with hydrogen peroxide and nitric acid, and slide-mounted with Naphrax<sup>TM</sup>. For each stream, 300-400 valves/season were counted at 1000 × magnification and identified to species. We



estimated the proportion of the assemblage that was acidobiontic (taxa optimally occurring at pH <5.5, (Van Dam et al. 1994) because pH increased with disturbance in our streams (Houser, unpublished data). We also focused on a subset of the assemblage that was considered motile taxa, which are often used to measure siltation because of their ability to move up through deposited sediment (Bahls 1993).

Macroinvertebrates. — We sampled macroinvertebrates 3 × per y (spring, summer, winter) from 2000 to 2002 using 4 Hester-Dendy (HD) multiplate units (each unit = 0.34 m<sup>2</sup>) spaced ~20 m apart in 8 of 12 streams (see Maloney and Feminella submitted, Table 1). HDs were deployed, allowed to colonize for approximately 8 wk, and then collected and returned to the laboratory for sample processing. In addition to HD samples, we took 2 D-frame sweep nets, 1 above and 1 below the HD stations, to sample general habitat during HD retrieval. We estimated the proportion of the macroinvertebrate assemblage that clinged to substrate as an indication of substrate availability. We also calculated the total number of taxa in the Dipteran family Chironomidae, which are generally tolerant to sedimentation (Culp et al. 1986, Shaw and Richardson 2001), and the number of taxa in the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)(EPT), which are generally considered sensitive macroinvertebrates (Barbour et al. 1999). We also calculated total taxa (richness) and Shannon diversity ( $H'$ ). We also used a region-specific multimetric index designed for streams within Georgia that is a modification of the Florida Stream Condition Index (Barbour et al. 1996), hereafter referred to as the Georgia Stream Condition Index as an index of biotic integrity (GASCI, GADNR 2002).



Fish. — Fish were sampled in 3 pool and 3 run habitats over a 100 m representative reach 3 × per y (spring, summer, winter) in 8 of 12 streams in 2003 using an electro-shock backpack (Smith-Root LR-24) and block seines above and below each habitat (see Maloney et al. submitted, Table 1). All fish, except for a few representative voucher specimens, were returned to their habitat following identification (species level) and measurement of standard length. Shannon diversity ( $H'$ ), total taxa (richness), and total number of collected individuals were calculated for the entire stream.

Metabolism calculations. — Stream metabolism was measured in 11 streams from summer 2001 through summer 2003 using a 1-station diurnal oxygen curve approach (Bott 1996, Houser et al. In Press);. Dissolved oxygen (DO) concentrations were measured at 15-minute intervals using YSI Model 6000 or 600 series sondes equipped with a YSI model 6562 DO probe. The sondes were deployed for 7 to 21 days in each stream each season (winter, spring, summer, and fall). Reaeration coefficients, stream velocity, and discharge were determined in the field using a simultaneous, steady-state injection of propane gas (volatile tracer) and a concentrated NaCl solution (conservative tracer) over reaches ranging in length from 45 m to 110 m (Genereux and Hemond 1992). Stream respiration and gross primary production were calculated from the DO data, using an approach similar to that used in a 2 station method by (Mulholland et al. (2001). The primary difference being that we calculated the rate of change in DO as the difference between consecutive DO readings. The net rate of oxygen change due to ecosystem metabolism was determined at 15 minute intervals



based on the change in DO concentration measured at the single station corrected for air-water O<sub>2</sub> exchange. The rate of air-water O<sub>2</sub> exchange was calculated using the reaeration coefficient and observed % DO saturation. Nighttime respiration was the sum of net ecosystem metabolism that occurred at night. Daytime respiration was determined by interpolating between respiration rates measured 1 hour before dawn and 1 hour after dusk. Total daily respiration was the sum of nighttime and daytime respiration rates over 24 hrs (from midnight to midnight). Daily GPP was the sum of the differences between interpolated daytime respiration rates and observed ecosystem metabolism.

Data analysis. — We used study average values for each physicochemical and biotic variable in multiple regression analyses to model relationships between each stream variable with historic (D44, R44) and contemporary (D99, R99) land use patterns. In addition, area of the catchment above our lowest sampling point (Area) was used as a natural basin characteristic in regression models. Analysis of multicollinearity (variable inflation factors, VIF) revealed no multicollinearity for explanatory variables. As a result of low sample size (7–12 streams) for each physicochemical or biotic variable, we only considered regression models with 3 or fewer variables. Land uses between the 2 time periods were often correlated (e.g.,  $D \underline{R}^2 = 0.63$ ,  $p = 0.03$ ), therefore models with multiple year land use (e.g., D44 and D99 in a 2-variable model) were removed from analysis. We then selected the model with the highest  $\underline{R}_{adj}^2$  value as the best model. All physicochemical and biotic variables were transformed to satisfy the conditions of linear regression analysis prior to analyses.



## Results

### Land use classification

Among study catchments, the proportion of disturbed ground on slopes >3% declined from a mean of 22.2% in 1944 (range 5.9 – 37.0%) to a mean of 5.6% in 1999 (range 0.1 – 10.6%), whereas the proportion of a catchment recovering on slopes >3% increased from a mean of 17.3% in 1944 (range 0.0 – 43.2%) to a mean of 21.6% (range 6.1 – 37.7%) in 1999 (Table 2). The proportion as unpaved roads decreased from a mean of 3.5% (range = 1.6 – 7.7) in 1944 to a mean of 2.5% (range = 0.89 – 4.3) in 1999. In general catchments were more disturbed (summation of D and R) in 1944 than 1999 (Figure 3).

### Physicochemical variables

Stream physicochemistry, other than bed instability, was related more to contemporary than historic land use patterns (Table 3). Results of the multiple regression analyses suggested a positive relationship between pH and D99, a negative relationship between DOC, BPOM, and CWD and D99, and a 3 variable relationship for SRP with catchment area, D99, and R99 (Table 3). Stream flashiness was positively related to D99, whereas bed instability was positively related to R44 and negatively with catchment area. Simple regression analyses with dependent physicochemical variables against D99 and D44 (the 2 best independent variables) indicated that streamwater pH was explained more so by D99 than D44, as was stream flashiness, whereas variation in CWD was negatively related to D99 (Figure 4).



### Stream biota

Multiple regression analyses results indicated a possible land use legacy on stream biota (Table 4). D99 and D44 were both significant predictors of the proportion of acidobiontic diatoms (negative relationship) and the proportion of motile diatoms (positive relationship). D44 best explained the % of the macroinvertebrate assemblage that cling to substrate (negative relationship), the number of EPT taxa (negative) and H-D density (negative), whereas D99 best explained variation in the number of Chironomidae taxa, total richness, and GASCI (all showed negative relationships, Table 4). Macroinvertebrate  $H'$  was best explained by a 2 variable model including D99 and R99. Fish measures were best explained by 1944 land use (Table 4). The number of fish collected decreased with increasing D44 as did richness and  $H'$ . The amount of variation in whole-stream respiration was best explained by D99 (negative), whereas variation in GPP was best explained by a 2 variable model containing catchment area and D44.

Simple regression analyses with dependent biotic variables against D99 and D44 suggested the proportion of acidobiontic diatoms was equally explained by D99 and D44, whereas the % macroinvertebrate that cling to habitat showed a stronger negative relationship with D44 than D99 (Fig. 5). Simple regression analyses also suggested that fish richness was best explained by D44, whereas whole stream respiration was explained by D99 (Fig. 5).



## Discussion

Legacies and stream physicochemistry. — In general, stream physicochemical variables were more related to contemporary landuse suggesting they recover from disturbance within a 60 y time period. However, bed stability was highly related to historic terrestrial disturbance suggesting it may take longer than 60 y to restabilize.

Stream flashiness is the hydrologic response of a catchment to a precipitation event. The rate of increase and subsequent decline in discharge during a storm event is dependent on overland and near-surface flow, which are products of rain intensity and catchment water retention characteristics. Contemporary military high-use catchments have decreased vegetation, which results in less water retention by vegetation and soils. Further, high-use areas experience soil compaction (Goran et al. 1983, Garten et al. 2003), which likely decreases soil infiltration rates resulting in higher levels of overland flow. The combined effects of decreased plant and soil retention and increased soil compaction likely led to increased overland flow during storm events and a subsequent flashier hydrologic response of streams in contemporary high-disturbed streams.

Our findings that streamwater chemistry was more strongly related to contemporary than historic land use is consistent with reports from other studies where changes in stream chemistry from catchment disturbance were short-term returning to pre-disturbed levels within 10 years (Likens et al. 1978, Lynch and Corbert 1991), but counter to the findings of Swank et al. (2001) who found increased streamwater  $\text{NO}_3$  levels 20 y after catchment disturbance. SRP was negatively related to both



contemporary D and R a result counter to many other studies (e.g., Byron and Goldman 1989, Lowrance et al. 1997). Military disturbance in our catchments results in an increase in clay-rich sediments in streams, which have a high affinity for SRP (Langmuir 1997). In fact, mean substrate particle size decreased with increasing catchment disturbance (Maloney et al. In press). Therefore, the increased proportion of clay in sediments and smaller bed particle size associated with catchment disturbance likely increased sorption of SRP to a greater extent in more highly disturbed catchments. We observed no relationship between any land use or drainage area with stream  $\text{NO}_3$  concentration. A likely explanation may be that any effect of disturbance on  $\text{NO}_3$  in these catchments was likely low and masked by the natural variability in  $\text{NO}_3$  concentrations caused by differential abundance of N-fixing vegetation (e.g., Myrica cerifera L., Alnus serrulata, B. G. Lockaby, Auburn University, unpublished data). The positive relationship between pH and contemporary land use was likely a result of higher buffering capacity in more disturbed streams (e.g.,  $\text{Ca}^{+}$  was slightly higher in catchments with D over 8%, mean 1.01 vs. those < 8 % D mean = 0.40, 2 tailed,  $p < 0.0001$ ), which possibly resulted from increased mobilization of base cations from disturbed catchments. Taken together, SRP,  $\text{NO}_3$ , and pH patterns suggest that present-day water chemistry was influenced mainly by contemporary catchment conditions and thus did not experience a land use legacy effect.

Coarse woody debris in our streams was more strongly related to contemporary than 60 y old land use which agrees with the findings of Hyatt and Naiman (2001), but is counter to many other reports of CWD having residence times of > 100 y (Wallace et al. 2001, May and Gresswell 2003). Prior to military purchase the majority of our sites



experienced intensive agriculture. We hypothesize that removal of CWD from channels during the intense agricultural period prior to military purchase may ultimately be responsible for this relationship. Although we have no data to support this, stream cleaning in agriculture catchments was a widespread and accepted form of channel maintenance in the past (Harmon et al. 1986). Removal of the majority of CWD would likely “reset” each stream’s CWD quantity to zero thus only CWD post-military purchase would be in the channels, thus catchments with higher CWD input due to contemporary land use effects would have more CWD in the stream channels. Benthic particulate organic matter (BPOM) has a residence time much shorter than CWD, a likely explanation of the negative relationship with contemporary disturbance level. Moreover, BPOM is retained by CWD (Bilby 1981, Wallace et al. 1995), thus the negative trends in both CWD and BPOM with contemporary disturbance levels are likely related.

Most bedload movement occurs during storm events, although in our contemporary high-disturbed streams bed movement was evident even at baseflow conditions (KOM personal observation). The sandy-bottom substrate of our stream channels originates from catchment sources, where sediment residence times can be decades to centuries old (Meade 1982, Trimble 1999). Many of the historic agriculture fields are currently forested areas with sediments eroded during the agricultural period continuing to migrate through ephemeral channels toward perennial streams. A likely explanation to our finding that bed instability was negatively related to catchment area and positively related to historic field cover may be that only recently has the sediment from historic agricultural practices reached and entered stream channels and may



continue to do so for decades with larger catchments experiencing a greater “lag” in the sediment pulse from historic disturbance.

Legacies and stream biota. — The relationship between biotic assemblages and disturbance legacies depends on disturbance intensity, recolonization sources, and species life history traits. If the historic disturbance is local and nearby recolonization sources remain, recolonization of the disturbed area may be rapid. However, if disturbance is widespread resulting in regional extinctions of fauna, then recovery and recolonization of the system may take decades or centuries to occur, if ever. Species-specific life history traits also may influence the effect of a disturbance legacy on contemporary biotic conditions. For example, species with short life spans and high turnover rates ( $> 1$  reproductive cycle  $y^{-1}$ ) should be able to recover from disturbance more rapidly than species with long life spans and low turnover rates ( $< 1$  reproductive cycle  $y^{-1}$ ). The Southeastern Plains of the U.S. has experienced widespread landscape and intensive catchment disturbance from historic agriculture and silviculture practices (Frost 1993, Kane and Keeton 1998), thus we would expect to see a legacy effect in relatively long-lived, low turnover biotic assemblages.

Diatom assemblages have short life spans and high turnover rates and as such should respond to environmental conditions rapidly, and therefore should be related more to contemporary than historic catchment conditions. Our findings that the proportions of acidobiontic taxa and motile taxa both were equally related to contemporary and historic land use, in part, agree with this hypothesis. These trends were likely driven by the dominance of taxa within the genus Eunotia (range 24 – 91%),



a genus comprised of mainly acidobiontic and non-motile diatoms. A likely explanation to the strong relationship between acidobiontic and motile diatoms with both historic and contemporary land use likely resulted from a combined effect of pH and stability, which were related to contemporary and historic land use, respectively. The higher pH levels in contemporarily disturbed catchments were above the optimal level for most of the acidobiontic taxa. Historically disturbed catchments had less stable streambeds, which may have created unsuitable habitat for the non-motile Eunotia taxa. In effect, we saw a coupling of a legacy effect (stability) with contemporary land use influence (pH) on present-day diatom assemblages.

Our results indicate a possible land use legacy effect on stream macroinvertebrates in that macroinvertebrate measures were related to both contemporary and historic land use. The % of the assemblage that clings to substrate, an indicator of stable substrate, was negatively related to the proportion of bare ground and unpaved road cover in a catchment in 1944. This negative relationship suggests a legacy effect on stable habitat, which is further supported by the positive relationship between bed instability and historic land disturbance from agriculture. A less stable bed may have reduced available habitat for taxa that require stable habitat. The number of taxa in the sensitive orders Ephemeroptera, Plecoptera, and Trichoptera were best explained by 1944 land use, suggesting these taxa experienced a legacy effect. Many of these taxa are sensitive to sediment and as such the decreased stability and increased inorganic suspended sediment concentrations associated with increased catchment disturbance ( $R^2 = 0.46$ ,  $p = 0.016$ ) may have reduced available habitat for these organisms as well. The number of Chironomidae taxa, assemblage diversity, and



Taxa Richness were all related to contemporary land use, however these indices may be related, in that chironomids made up >50% of the total number collected and ~40% of the total taxa collected. Chironomids, in general, are tolerant of sediment and the majority require BPOM as a food source (Merritt and Cummins 1996, Barbour et al. 1999). The negative relationship between streambed BPOM and contemporary land use may be, in part, responsible for the observed patterns in number of chironomids collected, total taxa, and assemblage diversity; less BPOM resulted in lower food availability for these taxa.

We also tested the efficacy of a regional multimetric index of integrity, GASCI, to discern contemporary and historic land use. The GASCI was related more so to contemporary land use than historic land use, indicating its potential for assessing contemporary stream impairment. The GASCI consists of 7 macroinvertebrate indices, 3 of which are highly dependent on the chironomid assemblage: Number Chironomidae Taxa, % Diptera, and Taxa Richness. Thus, the high relationship between 1999 D and GASCI may have been driven by the chironomid assemblage.

Fish abundance, richness, and diversity all were negatively related to historic land disturbance, indicating a land use legacy on the stream fish assemblage. The apparent legacy effect on fish assemblages likely resulted from the positive relationship between bed stability and historic land use. Reduced bed stability likely reduced available habitat for fish spawning, reducing the numbers of non-parental care fishes in the historically more disturbed streams. In fact, we previously reported opposite relationships between the dominant non-parental care fish (Pteronotropus euryzonus, negative) and the dominant parental-care fish (Semotilus thoreauianus, positive) with



contemporary land disturbance in our study sites (Maloney et al. in review). Further, decreased streambed stability may have decreased foraging efficiency via increased suspended solids of fish specializing on drift food resource (e.g., P. euryzonus), compared with those with a more omnivorous diet (e.g., S. thoreauianus).

Whole-stream rates of metabolism partly experienced a legacy effect with respiration being more related to contemporary land use, whereas GPP was more related to catchment size and historic land cover. The negative relationship between respiration and contemporary land disturbance likely resulted from lower amounts of labile carbon and CWD and associated debris dams ("hot spots" for benthic metabolism (Hedin 1990, Fuss and Smock 1996) in streams draining currently highly disturbed catchments. The negative relationship between GPP and catchment size indicates that primary productivity may be influenced mostly by stream size and to a lesser extent land use in our systems (Houser et al. In Press). However, there is weak evidence supporting a legacy effect on stream GPP. The unstable streambeds of historically disturbed catchments, an apparent legacy effect from pre-military land use, provided poor habitat for periphyton, as was evident from the diatom assemblage results.

The legacy effect on stream macroinvertebrate (% clingers, Number EPT) and fish assemblages (Number fish collected, Fish total taxa) was largely driven by one stream (BC2). This catchment was heavily disturbed in 1944 (D44 = 27.4, R44 = 3.4, and Roads = 3.3%) but became a contemporary reference stream (D99 = 0.33, R99 = 6.1, and Roads = 2.8%). Macroinvertebrate and fish assemblages often were depressed in this stream (Figs. 4 and 5) relative to other streams with little current and historical disturbance (e.g., HB, KM1), resulting in a non-significant relationship between



the aforementioned assemblage measures and present-day land use patterns. This result suggests that presence of a legacy effect in 1 of 7 (macroinvertebrate) or 8 (fish) streams may lead to misidentification of possible relationships.

#### Disturbance legacies and implications for stream ecology and conservation. —

Understanding legacy effects on contemporary ecosystem structure is an essential component of any ecological study. In addition to illuminate potential mechanisms behind outlier sample points, understanding legacy effects may also enhance insight of biotic interaction and associations with surrounding environment. In addition, identification of depauperate systems resulting from historic conditions within contemporarily low-disturbed catchments is an essential first step in identifying reference conditions in studies of biotic integrity. Finally, it is important to note our findings were comparative and do not imply cause and effect. Further studies that have both historical fauna and land use data are needed to promote understanding of legacies on stream ecosystems.



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Table 1. Site characteristics and years each stream sampled. Abbreviations: military land use - a = infantry/Ranger light impact, b = heavy machinery; Macro. = macroinvertebrates, CWD = coarse woody debris; 0 = 2000, 1 = 2001, 2 = 2002, 3 = 2003.

| Stream                 | Stream code | UTM                | Military land use | Stream order | Drainage area (km <sup>2</sup> ) | Year contemporary stream variables were sampled |        |      |            |                 |     |            |             |
|------------------------|-------------|--------------------|-------------------|--------------|----------------------------------|---|--------|------|------------|-----------------|-----|------------|-------------|
|                        |             |                    |                   |              |                                  | Diatoms   | Macro. | Fish | Metabolism | Water Chemistry | CWD | Flashiness | Instability |
| Bonham Tributary       | BC1         | 0710893N, 3588286E | a                 | 2            | 2.1                              | 1,2   | 0,1,2  | 3    | 1,2,3      | 0,1,2,3         | 2,3 | 0,1,2,3    | 3           |
| Bonham Tributary       | BC2         | 0710627N, 3588976E | a                 | 2            | 0.75                             | 1,2   | 0,2    | 3    | 1,2,3      | 0,1,2,3         | 2,3 | 0,2,3      | 3           |
| Hollis Branch          | HB          | 0717848N, 3583123E | a                 | 2            | 2.15                             | 1,2   | 1,2    | 3    | 1,2        | 1,2,3           | 2,3 | 1,2,3      | 3           |
| Kings Mill Creek Trib. | KM1         | 0720701N, 3600036E | a                 | 2            | 3.69                             | 1,2   | 1,2    | 3    | 1,2,3      | 0,1,2           | 2,3 | 1,3        | 3           |
| Kings Mill Creek Trib. | KM2         | 0719946N, 3599991E | a                 | 3            | 2.31                             | 1,2   | 1,2    | ---- | 1,2,3      | 0,1,2,3         | 2,3 | 1,2,3      | 3           |
| Lois Creek             | LC          | 0715377N, 3597908E | a                 | 2            | 3.32                             | 1,2   | 0,1,2  | 3    | 1,2,3      | 0,1,2,3         | 2,3 | 0,1,2,3    | 3           |
| Little Pine Knot Trib. | LPK         | 0719223N, 3585421E | b                 | 2            | 0.33                             | 1,2   | 0,1,2  | ---- | 1,2,3      | 1,2,3           | 2,3 | ----       | 3           |
| Sally Branch Trib.     | SB1         | 0716349N, 3585850E | b                 | 3            | 5.43                             | ----  | 0,1,2  | ---- | ----       | 0,1             | 2   | ----       | ----        |
| Sally Branch Trib.     | SB2         | 0716808N, 3584787E | b                 | 2            | 1.23                             | 1,2   | ----   | 3    | 1,2,3      | 1,2,3           | 2,3 | 1,2,3      | 3           |



|                    |     |                       |   |   |      |     |      |      |       |       |     |       |      |
|--------------------|-----|-----------------------|---|---|------|-----|------|------|-------|-------|-----|-------|------|
| Sally Branch Trib. | SB3 | 0716673N,<br>3584684E | a | 1 | 0.72 | 1,2 | ---- | 3    | 1,2,3 | 1,2,3 | 2,3 | 1,2,3 | 3    |
| Sally Branch Trib. | SB4 | 0716005N,<br>3584889E | b | 1 | 1    | 1,2 | ---- | 3    | 1,2,3 | 1,2,3 | 2,3 | 1,2,3 | 3    |
| Sally Branch Trib. | SB5 | 0714935N,<br>3585249E | b | 2 | 1.27 | 1   | 1,2  | ---- | 1,2,3 | 1,2,3 | 2   | ----  | ---- |

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Table 2. Results of land use classifications. Each column represents the percentage of each land use within each catchment. Disturbed and recovering land were restricted to those on slopes > 3%. Stream abbreviations are given in Table 1.

| Stream | 1944 coverage |            |       | 1999 coverage |            |       |
|--------|---------------|------------|-------|---------------|------------|-------|
|        | Disturbed     | Recovering | Roads | Disturbed     | Recovering | Roads |
| BC1    | 26.46         | 15.29      | 5.97  | 7.15          | 12.56      | 3.31  |
| BC2    | 27.40         | 3.37       | 3.27  | 0.33          | 6.07       | 2.82  |
| HB     | 5.88          | 43.19      | 2.18  | 4.57          | 26.56      | 2.05  |
| KM1    | 8.59          | 13.27      | 2.97  | 1.46          | 14.40      | 3.16  |
| KM2    | 18.60         | 15.19      | 1.57  | 0.14          | 13.05      | 1.68  |
| LC     | 10.29         | 35.10      | 1.98  | 2.78          | 22.25      | 0.89  |
| LPK    | 22.58         | 0.00       | 4.80  | 9.66          | 29.23      | 1.60  |
| SB1    | 24.07         | 14.66      | 2.81  | 5.80          | 22.42      | 2.58  |
| SB2    | 25.74         | 28.22      | 1.55  | 9.09          | 32.16      | 1.40  |
| SB3    | 25.73         | 26.02      | 4.05  | 4.76          | 37.71      | 3.36  |
| SB4    | 37.01         | 12.46      | 3.57  | 10.57         | 22.20      | 3.08  |
| SB5    | 33.43         | 0.85       | 7.67  | 10.40         | 20.17      | 4.25  |



Table 3. Results of multiple regression analyses of measures of stream physicochemistry at two different time periods. D99, R99, D44, R44 = proportion of catchment the was disturbed in 1999, recovering in 1999, disturbed in 1944, and recovering in 1944, respectively. Area = drainage area above lowest sampling point in stream. SRP = soluble reactive phosphorus, DOC = dissolved organic carbon, BPOM = benthic particulate organic matter, CWD = channel coarse woody debris, n.s. = no significant models. Numbers in parentheses denote beta weights from regressions.

| Variable        | Land use / Catchment area               | $R^2_{adj}$ | F     | p      |
|-----------------|---|-------------|-------|--------|
| pH              | D99 (0.11)                              | 0.55        | 13.00 | 0.0057 |
| SRP             | Area (0.38), D99 (– 0.13), R99 (– 0.06) | 0.89        | 29.68 | 0.0001 |
| NO <sub>3</sub> | n.s.                                    | ----        | ----  | ----   |
| DOC             | D99 (– 0.13)                            | 0.40        | 7.07  | 0.0289 |
| BPOM            | D99 (– 0.01)                            | 0.62        | 17.30 | 0.0024 |
| CWD             | D99 (– 0.01)                            | 0.79        | 38.77 | 0.0002 |
| Flashiness      | D99 (0.02)                              | 0.54        | 10.65 | 0.0141 |
| Instability     | R44 (0.07), Area (– 0.99)               | 0.77        | 14.54 | 0.0050 |



Table 4. Results of multiple regression analyses of measures of stream biotic communities at two different time periods. D99, R99, D44, R44 = proportion of catchment the was disturbed in 1999, recovering in 1999, disturbed in 1944, and recovering in 1944, respectively. Area = drainage area above lowest sampling point in stream. GPP = gross primary productivity, EPT = number macroinvertebrate taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera, GASCI = Georgia Stream Condition Index. Numbers in parentheses denote beta weights from regressions.

| Variable                  | Land use / Catchment area   | $R^2_{adj}$ | F     | p      |
|---------------------------|-----------------------------|-------------|-------|--------|
| <b>Diatoms</b>            |                             |             |       |        |
| % Acidobiontic            | D99 (– 0.04)                | 0.72        | 26.50 | 0.0006 |
|                           | D44 (– 0.01)                | 0.70        | 24.80 | 0.0008 |
| % Motile taxa             | D99 (0.02)                  | 0.41        | 10.86 | 0.0205 |
|                           | D44 (0.01)                  | 0.40        | 7.64  | 0.0220 |
| <b>Macroinvertebrates</b> |                             |             |       |        |
| % Clingers                | D44 (– 0.55)                | 0.91        | 59.85 | 0.0006 |
| No. chironomid taxa       | D99 (– 0.72)                | 0.64        | 11.79 | 0.0186 |
|                           | D99 (– 0.63), Area (0.79)   | 0.75        | 10.10 | 0.0273 |
| EPT                       | D44 (– 0.28)                | 0.78        | 22.41 | 0.0052 |
|                           | D99 (– 0.66)                | 0.70        | 15.27 | 0.0113 |
| Shannon H'                | D99 (– 0.04)                | 0.41        | 5.18  | 0.0719 |
|                           | D99 (– 0.06), R99 (0.02)    | 0.65        | 6.56  | 0.0546 |
| Richness                  | D99 (– 1.40)                | 0.53        | 7.85  | 0.0379 |
| Density                   | D44 (– 29.7)                | 0.55        | 8.38  | 0.0340 |
| GASCI                     | D99 (– 0.83)                | 0.92        | 74.43 | 0.0003 |
| <b>Fish</b>               |                             |             |       |        |
| Number collected          | D44 (– 1.11)                | 0.61        | 11.74 | 0.0140 |
| Richness                  | D44 (– 0.14)                | 0.81        | 31.67 | 0.0013 |
| Shannon H'                | D44 (– 0.03)                | 0.80        | 29.28 | 0.0016 |
| <b>Metabolism</b>         |                             |             |       |        |
| Respiration               | D99 (– 0.20)                | 0.44        | 8.86  | 0.0155 |
| GPP                       | Area (– 0.75), D44 (– 0.05) | 0.56        | 7.38  | 0.0153 |



Figure 1. Map showing locations of study catchments (depicted by polygons) within Fort Benning Military Installation, GA. Dotted line in left figure in top panel represents the Chattahoochee River, which separates Alabama (AL) and Georgia (GA). Numbers in the right figure identify watersheds on the same stream (e.g., 1 and 2 on the King's Mill Creek represent King's Mill Creek Tributaries 1 and 2, KM1 and KM2, respectively).

Figure 2. Aerial photographs of catchment SB3 in 1944 and 1999. D44 represents recently abandoned fields in 1944, R44 recovering areas in 1944, D99 disturbed areas in 1999, R99 recovering areas in 1999.

Figure 3. Percentage of each catchment disturbed on slopes  $>3\%$  (D), and recovering on slopes  $> 3\%$  (R) for 1944 and 1999. Ranked in order of 1999 disturbance.

Figure 4. Simple regressions between channel coarse woody debris (CWD), pH, and stream flashiness with 1944 and 1999 disturbed land. Open symbol in submerged CWD and pH analysis indicates an outlier catchment (BC1) that was removed from analysis.

Figure 5. Simple regressions between representative stream biotic measures and the proportion of disturbed land in a catchment during 1944 and 1999. n.s. = non-significant relationship.



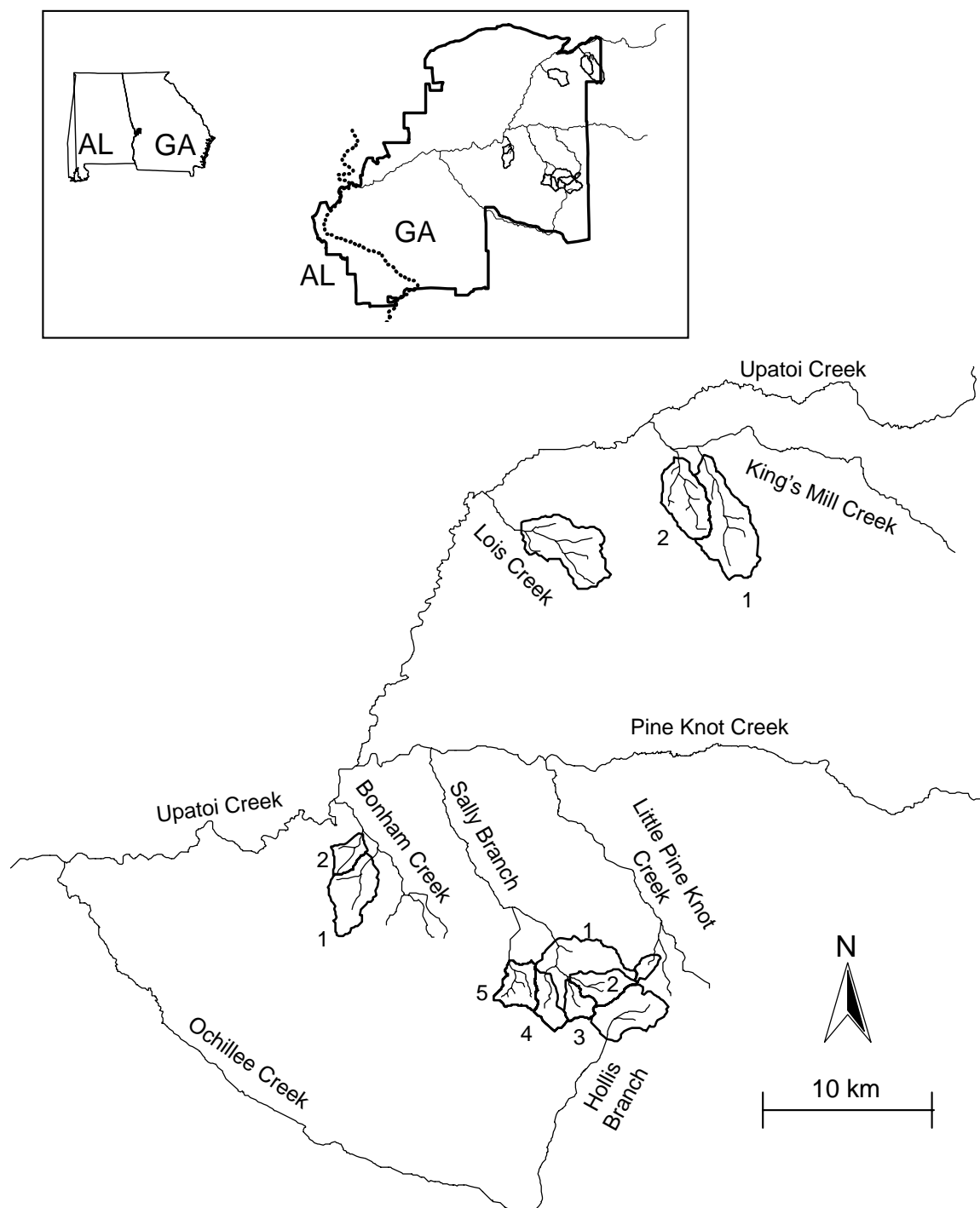


Figure 1.



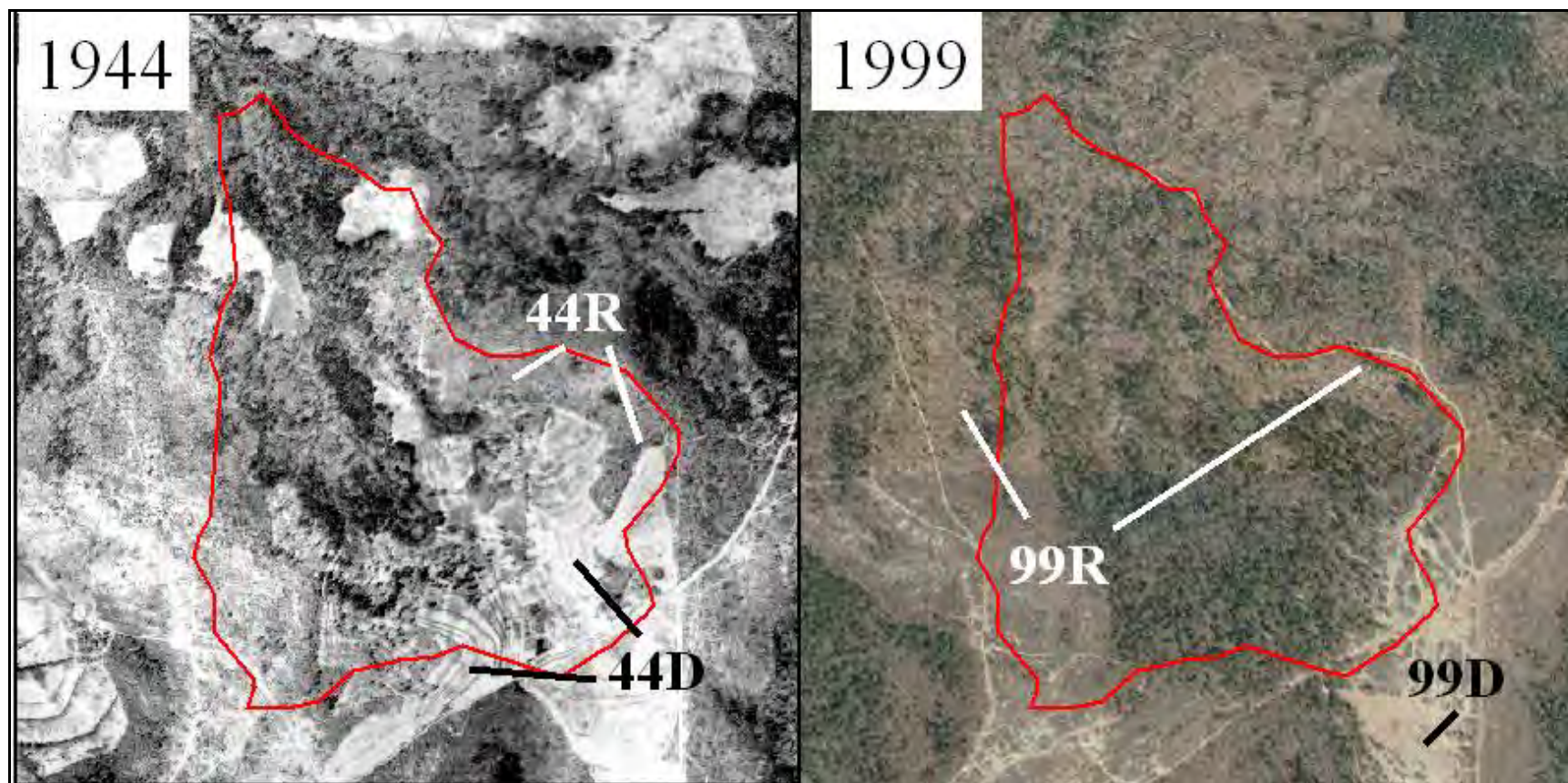


Figure 2.



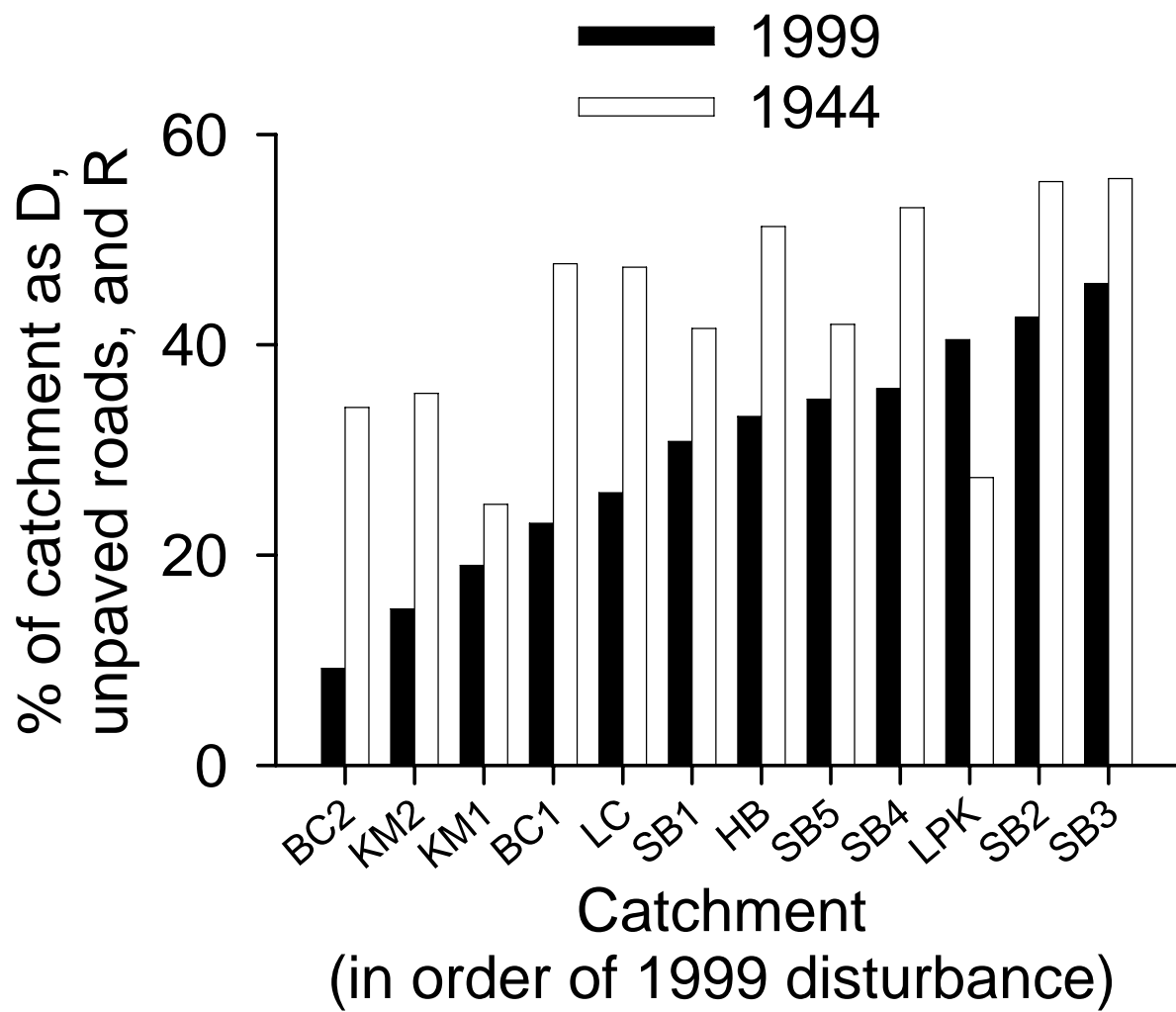


Figure 3.



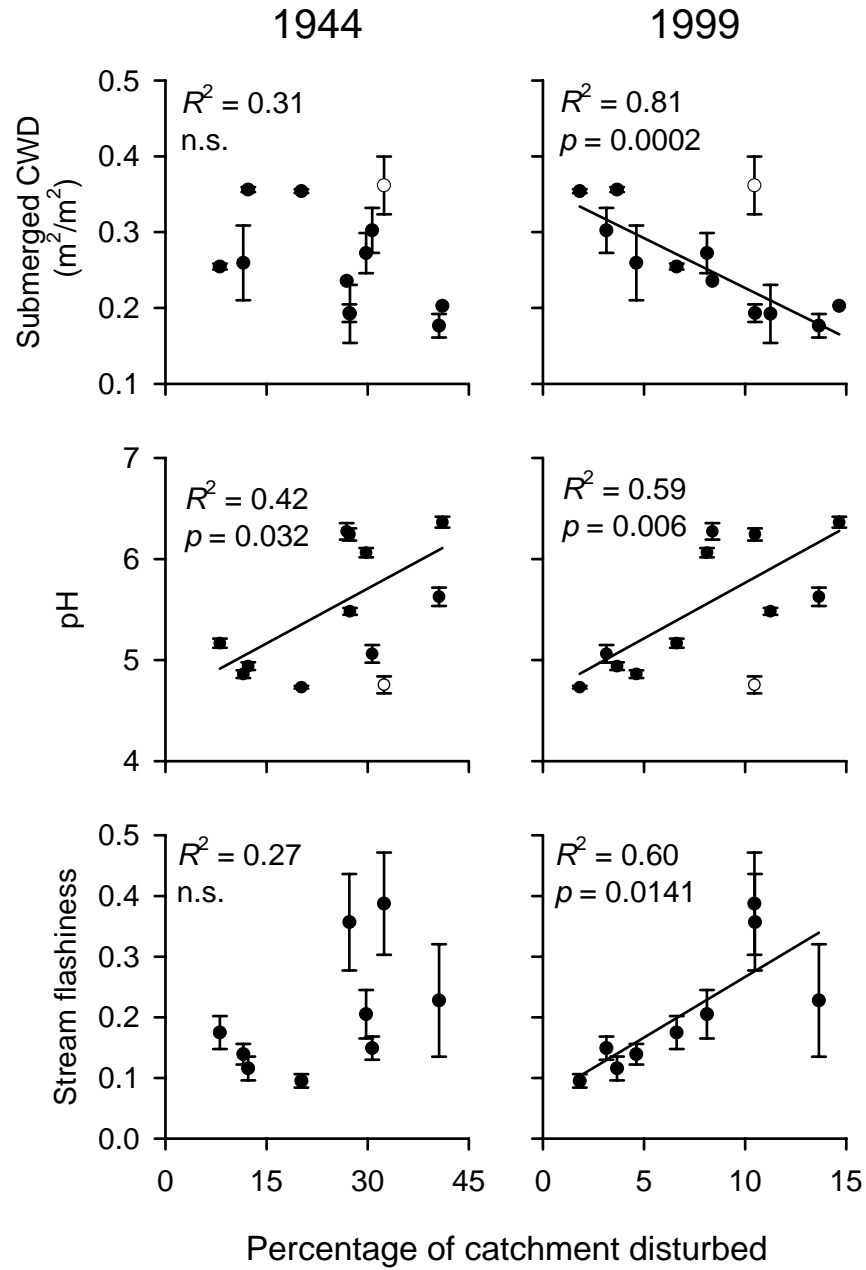


Figure 4.



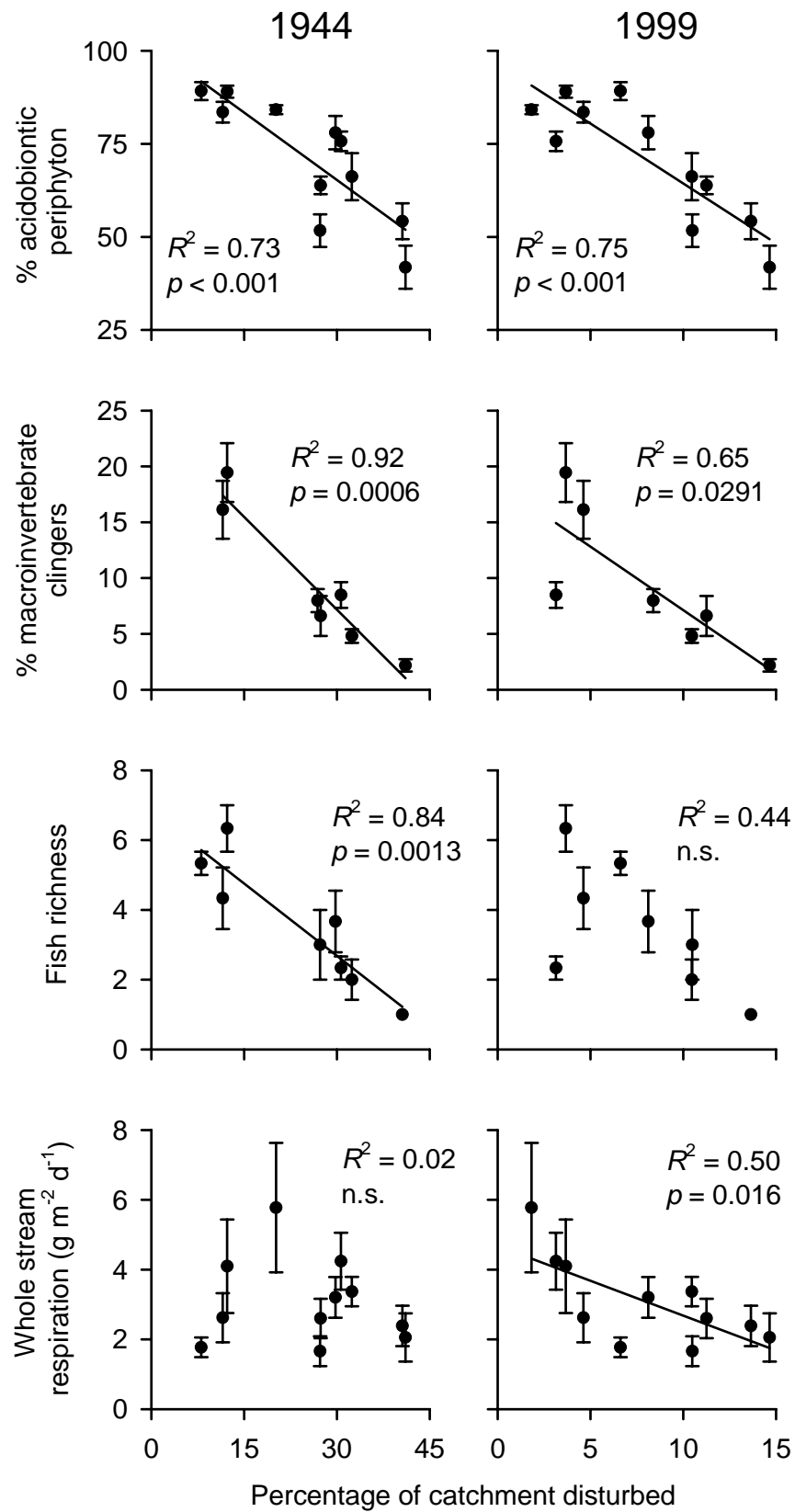


Figure 5.



## Section 8 of Final report of Indicators of Ecological Change

This section consists of four scientific papers:

Appendix A: Dale, V.H., Beyeler, S.C., and Jackson, B. 2002. Understory indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. **Ecological Indicators** 1(3): 155-170.

Appendix B: Peacock, A. D., S. J. MacNaughton, J. M. Cantu, V. H. Dale and D. C. White. 2001. Soil microbial biomass and community composition along an anthropogenic disturbance gradient within a longleaf pine habitat. **Ecological Indicators** 1(2):113-121.

Appendix C: Druckenbrod, D.L., and V.H. Dale. In preparation. Experimental response of understory plants to mechanized disturbance in longleaf pine forests at Fort Benning, Georgia. **Ecological Applications**

Appendix D: Dale, V.H., Peacock, A., C. Garten, and E. Sobek. In preparation. Contributions of soil, microbial, and plant indicators to land management of Georgia pine forests.

Together these studies show that characteristic of understory vegetation, soil microbiology and soil carbon are important indicators of ecological change at Fort Benning.



## Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA

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### Abstract

Environmental indicators for longleaf pine (*Pinus palustris*) ecosystems need to include some measure of understory vegetation because of its responsiveness to disturbance and management practices. To examine the characteristics of understory species that distinguish between disturbances induced by military traffic, we randomly established transects in four training intensity categories (reference, light, moderate, and heavy) and in an area that had been remediated following intense disturbance at Fort Benning, GA. A total of 134 plant species occurred in these transects with the highest diversity (95 species) in light training areas and the lowest (16 species) in heavily disturbed plots. Forty-seven species were observed in only one of the five disturbance categories. The variability in understory vegetation cover among disturbance types was trimodal ranging from less than 5% cover for heavily disturbed areas to 67% cover for reference, light, and remediated areas. High variability in species diversity and lack of difference in understory cover led us to consider life-form and plant families as indicators of military disturbance. Life-form successfully distinguished between plots based on military disturbances. Species that are Phanerophytes (trees and shrubs) were the most frequent life-form encountered in sites that experienced light infantry training. Therophytes (annuals) were the least common life-form in reference and light training areas. Chamaephytes (plants with their buds slightly above ground) were the least frequent life-form in moderate and remediation sites. Heavy training sites supported no Chamaephytes or Hemicryptophytes (plants with dormant buds at ground level). The heavy, moderate, remediated, and reference sites were all dominated by Cryptophytes (plants with underground buds) possibly because of their ability to withstand both military disturbance and ground fires (the natural disturbance of longleaf pine forests). Analysis of soils collected from each transect revealed that depth of the A layer of soil was significantly higher in reference and light training areas which may explain the life-form distributions. In addition, the diversity of plant families and, in particular, the presence of grasses and composites were indicative of training and remediation history. These results are supported by prior analysis of life-form distribution subsequent to other disturbances and demonstrate the ability of life-form and plant families to distinguish between military disturbances in longleaf pine forests. © 2002 Elsevier Science Ltd. All rights reserved.

**Keywords:** Longleaf pine; Indicators; Life-form; Disturbance; Family diversity

### 1. Introduction

Resource managers need a basic understanding of potential effects of human activity on ecological conditions. Human activity may influence a variety

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of ecological attributes including the presence of species, populations, and communities as well as the occurrence, rate, or scale of processes (Angermeier and Karr, 1994). Understanding the implications of anthropogenic disturbances on an ecological system is complicated by variability in ecological response. Identification of indicators which capture key ecological responses to human actions provides a useful tool for improving understanding of ecological effects and for monitoring and management.

Longleaf pine (*Pinus palustris*) forests are a system in which understanding effects of anthropogenic activity is necessary for resource management. Forests of the southeastern United States comprise a landscape that has experienced significant anthropogenic activity in the form of land development, resource utilization, and changes to the natural disturbance regimes. Anthropogenic activity within a landscape is typically expressed as a complex gradient of altered ecological components and changes in natural disturbance dynamics and succession patterns (Guntenspergen and Levenson, 1997). Prior to European settlement, longleaf pine forests covered 25–35 million hectares (ha) of the southeastern Coastal Plain landscape (Frost, 1993). By the 1900s, less than 10% of the original stands remained (Frost, 1993). Today only two million hectares of the pre-settlement forest endures (Quicke et al., 1994). The loss and degradation of the longleaf pine forest is mainly attributed to land-use change, timber harvest, and fire suppression (Haywood et al., 1998; Gilliam and Platt, 1999). Since the longleaf pine forests are a fire-adapted system, it is the absence of regular light ground fires that is a disturbance to these forests. Fires reduce the growth of hardwoods into the overstory.

The need for a clear understanding of human impacts on longleaf pine forests takes on even greater importance when considering the fact that much of the remaining longleaf pine forest supports not only critical ecological processes but also a multitude of ecosystem services (Noss, 1989). For example, the federally endangered red-cockaded woodpecker (*Picoides borealis*) is a nonmigratory bird endemic to the longleaf pine forests in the southeastern United States. A prime cause of decline in red-cockaded woodpecker populations is the loss and degradation of longleaf pine forests. Reduction of the woodpecker population would also induce decline of the 23 species

that inhabit holes in living trees uniquely created by these birds (Dennis, 1971).

One way to maintain the diverse ecological services of the longleaf pine forests entails reducing the amount of hardwood in-growth that, at first, compromises the understory and, eventually, alters overstory composition. As the hardwood trees grow into the canopy, the red-cockaded woodpeckers and other species unique to these forests tend to abandon the stands (Noss, 1989). Thus, the status of the understory composition and structure is a critical indicator of future condition of the longleaf pine forest (James et al., 2001). Unfortunately, the attributes and dynamics of this forest layer are not well-known, particularly for those systems that do not support the wire grass (*Aristida stricta* Michx.) community typical of the understory of some longleaf pine forests (Noss, 1989). Although an understanding of the cause and effect relationships of human modifications and alterations of longleaf pine systems is developing (Platt et al., 1988a,b; Frost, 1993; McCay, 2000), much still remains to be learned about human impacts on the understory in order to predict how human activities affect the ecological system.

Approximately, 75% of the longleaf forest is in private ownership serving a diversity of purposes including recreation and resource extraction. The remaining land is public. Almost without exception, the larger patches of longleaf pine forest are under federal ownership, a significant portion of which is on Department of Defense (DoD) lands (Walker, 1999). These large patches of intact forest best represent the ecological condition of the longleaf pine forest and tend to support the highest number of native species (Noss, 1989).

The longleaf pine stands on military installations are not only important forest reserves; they also provide suitable terrain for military training. In order to continue to meet the joint but seemingly incongruous needs of habitat reserves and military training, a means to monitor impacts of training should be developed and implemented. A critical challenge is to construct management procedures based on cost-effective monitoring plans that allow multiple land-use activities to take place while at the same time maintaining the ecological services of natural resources for the majority of the installation. There is a need on most military lands for the designation of sacrifice areas where training activities involving tracked vehicles and range practices must take place at the expense of ecological



integrity. However, some attempts are made to minimize impacts through soil conservation measures. In contrast, dismounted training that also occurs on the installation appears to have minimal immediate impact on the forest stands. Differences subsequent to military foot traffic occur in soil infiltration rates, erosion, above ground biomass, and litter (Whitecotton et al., 2000). Yet, effects of foot traffic on the understory vegetation and over the long-term are not well-known.

Our perspective is that a suite of indicators ranging from microbiologic to landscape metrics is necessary to capture the full spatial, temporal, and ecological complexity of impacts that should be measured. Potential indicators should be considered in a spatially hierarchical fashion and for all gradients deemed important at a site. Placing potential indicators on a spatial axis (e.g. Fig. 1) provides a means to ensure that information is considered across spatial scales. Alternatively, it is important to include indicators that encompass the diversity of responses over time (so that one is not just measuring short-term responses of the system). In a similar fashion, as depicted in this figure, all major gradients should be included in the analysis of potential indicators. Thus, it is useful to

consider the representativeness of indices across major physical gradients (e.g. soils, geology, land-use) and across gradients in disturbance regimes.

This study is part of a larger project designed to investigate indicators that would be useful to augment current sampling regimes at military bases and typical of other actively managed sites. Current ecological monitoring on military lands, the land condition trend analysis (LCTA) (Diersing et al., 1992), does not incorporate the diversity of indicators that are necessary for monitoring changes and responses to land as shown in Fig. 1. We hypothesized that understory conditions are a key element in the suite of indicators that can reflect differences in military training intensity. While some of the indicators from the proposed suite are designed to measure changes that occur over the long-term, understory vegetation is the element representing ecological changes that may occur over a few years to decades. Before such a suite can be adopted, it is necessary to evaluate how effectively the component indicators represent changes and susceptibility of ecological systems to military training. The purpose of this paper is to examine the ability of understory vegetation to indicate differences in disturbance regimes.

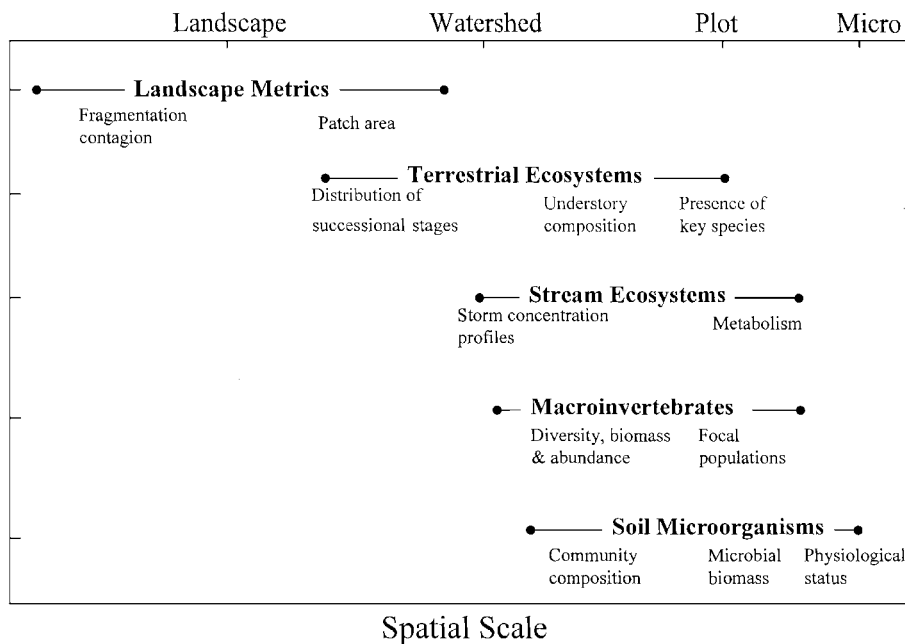


Fig. 1. Spatial hierarchical overlap of a suite of ecological indicators for Fort Benning, GA.



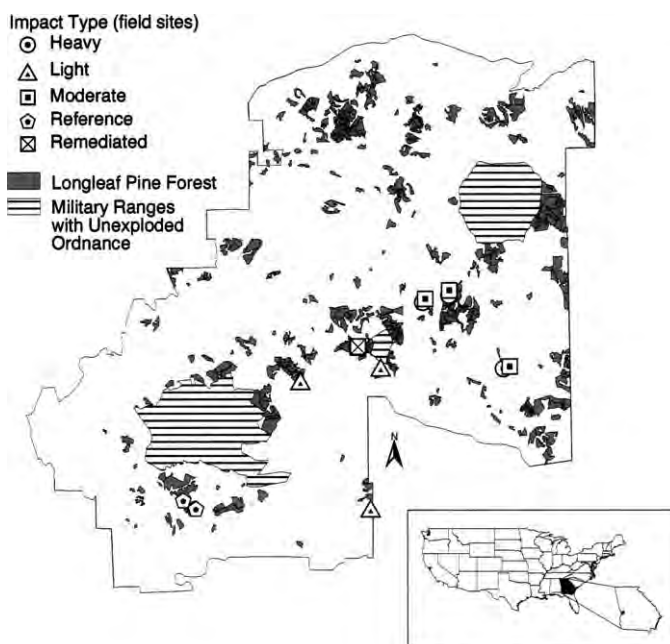


Fig. 2. Map of Fort Benning showing the location of field sites in relation to longleaf pine forest and military ranges that contain unexploded ordnance. Not that several sites are so close that the symbols overlap. The inset depicts the location of Fort Benning in western Georgia and Georgia in the southeastern United States.

## 2. Study site

The study was conducted at the Fort Benning Army Installation which occupies 73,503 ha in Chattahoochee, Muscogee, and Marion Counties of Georgia and Russell County of Alabama (Fig. 2). The climate at Fort Benning is humid and mild with rainfall occurring regularly throughout the year. The warmest months are July and August with average daily maximum and minimum temperatures of 37 and 15 °C, respectively. The coldest months are January and February with an average daily maximum and minimum temperature of 15.5 and –1 °C, respectively. Annual precipitation averages 105 cm with October being the driest month.

Fort Benning is located within the southern Appalachian Piedmont and Coastal Plains and is considered part of the southeastern Mixed Forest Province of the subtropical division (Bailey, 1995). The northern boundary of the installation lies along a transition zone between the Piedmont and Upper Coastal Plain. The installation is comprised of five major geologic

formations: undifferentiated alluvium and mixed terrace deposits; Cusseta formation, which is mostly micaceous sand; Bluffton formation, a layered micaceous sand; Tuscaloosa formation; and the Eutaw formation (Roemer et al., 1994). The soils are constituted of a combination of clay beds and weathered Coastal Plain material as well as alluvial deposits from the Piedmont. Eight soil associations form the majority of the soil on the installation. Lakeland–Troup, Orangeburg–Dothan–Ailey, and Raanoke–leaf soil associations occupy the higher elevations. Bibb–Chewacla–Rains, Ochlocknee–Toccoa, Augusta–Ochlocknee, and Susquehanna–Duplin–Esto are located on the alluvial flood plains and terraces. Undifferentiated rough gullied land occurs in the southeast portion of the installation (Elliot et al., 1995).

Historically, the land was cleared and actively farmed first by native American and later by European settlers (Kane and Keeton, 1998). Fort Benning was established in 1918, and all farming stopped as landowners were relocated (which occurred up to 1945). Military training ensued for the following



eight decades with heavy training land impacts occurring only in selected portions of the installation. Some timber harvesting and thinning continued, and the longleaf pine forests were subjected to regular low level fires for management purposes (Jack Greeley, personnel communication 1999, Fort Benning, GA).

Fort Benning contains several unique environmental features probably because the Fort Benning army installation was protected from farming and urban development which occupies much of the surrounding region. The presence of the federally-listed red-cockaded woodpecker is one reason why this study focused on the longleaf pine ecosystem. However, there are other rare species and habitats at Fort Benning, including the gopher tortoise (*Gopherus polyphemus*) and relict trillium (*Trillium reliquum*). Minimizing conflicts between the rare species and military land-use is a key goal of land management activities at the installation.

The presence of natural vegetation enables realistic training scenarios involving cover, concealment, or line-of-sight firing constraints. In order that Fort Benning can meet its mission needs now and into the future, the natural resources that provide the training context must be managed such that they are ecologically sustainable. With appropriate measurements and management, the retention of the training mission will also protect rare habitats and species at Fort Benning and other military installations.

The installation is a center for both dismounted and mechanized training, and, therefore, land-use focuses on military training (Waring et al., 1990). Maneuver areas are subject to a range of training activities such as dismounted infantry, mechanized forces, munitions detonation, biovac sites, landing strips and pads, and drop zones for airborne training (USAIC, 2001). Impacts of maneuver training activities on natural resources vary from direct removal or damage of vegetation, digging activities, ground disturbance from vehicles, soil compaction, soil erosion, and sedimentation. The degree and extent of the impacts of training activities depend on the type of training activity, time of year, intensity (e.g. the number of soldiers or vehicles per area per unit time), and how frequently the area is exposed to training activity. Further, different types of training typically occur irregularly over the landscape, and in many cases overlap, creating localized gradients of impacts. This study was limited to maneuver training

areas and, thus, does not include firing ranges, ordnance impact areas, or cantonment areas. Our goal was to develop valid and repeatable measures of impacts of training on understory of longleaf pine forests.

### 3. Experimental design

Study site locations were on land suitable for longleaf pine growth. Determination of potential site locations was achieved through a combination of existing forest stand information (Bob Larimore, personal communication, 1999, Fort Benning, GA) and county soil surveys of the United States Department of Agriculture Natural Resource Conservation Service (USDA NRCS, 1924, 1983, 1993, 1997). We overlaid an image of the United States Forest Service forest stand classification onto USDA NRCS soil maps for the area of land within the Fort Benning boundary. A final map was then created depicting locations of soils associated with longleaf pine within the installation boundary, and study sites were selected from those areas. Longleaf pine stands currently comprise approximately 5800 ha of the total area of Fort Benning (USAIC, 2001). Soils favorable to the establishment and growth of longleaf pine make up approximately 65,900 ha (about 90% of the total area).

The study was designed using a stratified sampling methodology. The sampling sites were blocked into five training intensity categories: reference, light, moderate, heavy, and remediation. Reference areas experience little to no training activities and are often in exclusion zones around firing ranges. Light impact areas are limited to dismounted training and individual orienteering activities. Moderate impact areas occur adjacent to tank training zones and are, thus, exposed to some tracked vehicle maneuvers, as well as limited vehicle and infantry traffic. Heavy impact areas are used exclusively for wheeled and tracked vehicle training exercises. The classification of each site was primarily based on historical records of training activity; however, due to the variability of training intensity over space, final site selection was achieved through field reconnaissance and discussions with the Fort Benning natural resource personnel.

The remediation area is located in the uplands of the McKenna Drop Zone that was cleared in 1988 and subsequently rehabilitated (but was not used for



training). It is currently off-limits to military training and testing. Revegetation efforts involved liming, fertilizing, and seeding with mixtures of grasses and legumes selected to increase vegetative cover and reduce run-off rates [e.g. giant reed (*Arundo donax*), bermuda grass (*Cynodon dactylon*), little bluestem (*Adropogon scoparius*), maidencain (*Panicum hemitomom*), pensacola bahiagrass (*Paspalum notatum*), alamo switchgrass (*Panicum virgatum*), weeping lovegrass (*Eragrostis curvula*), lespedeza sericea (*Lepedeza cuneta*, var. *Sericia*) and lespedeza interstate (*Lepedeza cuneta*, var. *Interstate*)].

Three transects were located in each of the reference, light, moderate, and heavy training classifications, and two transects were located in the remediation areas. Each of the 14 transects was established at a random distance and direction from a selected location.

Five circular plots were established along each transect at intervals of 15 m between the centers. The circular plot size of 5 m radius was determined based upon a species–area curve constructed for the reference site using the technique described by Barbour et al. (1980). At that size plot, 31 understory species occurred. Within each plot, all species of understory vegetation (less than one meter in height) were identified and assigned a cover class using a modified Braun-Blanquet (1932) cover system (based on Clarke, 1986) (Table 1).

Bråkenhielm and Qinghong (1995) have demonstrated that visual estimates provide the most accurate, sensitive, and precise measure of vegetation cover compared to point frequency and subplot frequency

methods. Thus, visual estimates of understory cover were used in this study. We came to a clear agreement in the field as to the appearance of each cover class. Individual species cover scores could not exceed 100%; however, cumulative cover scores for all species associated with an individual plot could be larger than 100%. All species were also classified using Raunkiaer's life-form classification system (Kershaw and Looney, 1985) based on the height of perennating buds.

Understory vegetation included all shrubby and herbaceous vegetation as well as trees under 5 cm diameter at breast height (DBH). In addition, canopy cover, canopy species, size of trees greater than 5 cm DBH, evidence of human disturbance, and depth of soil A horizon were recorded for each plot. The soil depth was meant to provide a quantitative measure of disturbance. In order to establish maximum stand age, we obtained two tree cores from each of the four largest trees in the immediate vicinity of each transect.

All species identification and characteristic descriptions were based on Godfrey (1988) and Radford et al. (1968). In a few cases plants could only be identified at the genus level. Understory oak had great plasticity, and distinguishing between saplings of the eight oak species was difficult. In addition, three distinct species of *Prunus* were observed, but due to a lack of a terminal inflorescence, two of the species were unidentifiable. Finally, one species of *Desmodium* was identified as clearly distinct from all other *Desmodium* species found within the study plots but was bearing no fruit, therefore rendering it impossible to identify.

Statistical analysis was performed to test for differences between the training intensities. Analysis of variance (ANOVA) was used to examine for differences in the mean cover scores for all species found within the plots (i.e. zeroes were eliminated). One-way ANOVA and Cochran–Mantel–Haenszel statistical test (Cochran, 1954; Mantel and Haenszel, 1959; Mantel, 1963) were conducted to see if there were differences in the frequency of cover ranks by life-form within a training category. We note that the ANOVA is asymptotically equivalent to the Kruskal–Wallis test. Then a two-way ANOVA was conducted to examine for differences in cover ranks considering both life-form and training category. The cover ranks were normally distributed by training category except for the heavy training sites.

Table 1

Key of the modified Braun-Blanquet (1932) cover classification system<sup>a</sup>

| Cover-abundance class | Species cover and distribution characteristic |
|-----------------------|---|
| 0                     | No plants present                             |
| 1                     | Less than 1% cover; 1–5 small individuals     |
| 2                     | Less than 1% cover; many small individuals    |
| 3                     | Less than 1% cover; few large individuals     |
| 4                     | 1–5% cover                                    |
| 5                     | 5–12% cover                                   |
| 6                     | 12–25% cover                                  |
| 7                     | 25–50% cover                                  |
| 8                     | 50–75% cover                                  |
| 9                     | 75–100% cover                                 |

<sup>a</sup> Modified from Table 2.3; Clarke (1986).



#### 4. Results

Highest understory plant species diversity occurred in light training sites and reference areas which also contained the oldest trees (Table 2). However, richness was also high in moderate training and remediation areas. Both diversity and understory plant cover were lowest in the heavy training areas which did not have a developed overstory. The moderate training areas had about two-thirds the amount of understory cover as did reference, light training, and remediation sites, and understory cover for those three areas was not distinguishable. Tree cover was highest in reference and light training areas, absent in heavy training areas, and very low in moderate training and remediation sites (Table 2).

A total of 134 understory plant species representing 36 families were identified in different training regimes at Fort Benning (see Appendix A). Many species had high variation in cover over all the training types, and we were unable to separate training types by species using multivariate analyses. Most species contributed an average of less than 1% cover. Little bluestem (*Andropogon scoparius*) had the highest mean cover (2.64%). Three awn grass (*Aristida oliganthum*) was the only species that occurred in all five training categories. Eight species were found only in reference and light training sites. Some species were found in only one training type: 11 in reference sites, 13 in light training, 14 in moderate training, and 4 in remediated sites. However, there were no species that occurred only in heavy training sites. Moderate training supported eight species which also occurred in sites with heavy training.

Families that contributed greater than 1% cover to the understory also differed by training category (Fig. 3). Grasses (Graminae) had the most cover for all categories. The heavy training had very little grass cover (2%), but grass cover exceeded 45% for moderate and reference areas and was greater than 75% for remediated areas. The reference sites had more than 30% cover of composites (Asteraceae) compared to 17% composite cover for light training areas and less than 5% for other training categories. Light training areas had the broadest taxonomic representation with 10 families contributing more than 1% cover as compared to one family (Graminae) for heavy training, four for moderate training, and six each for the reference and remediated sites.

Raunkiaer's life-form accounted for some differences between disturbances (Fig. 4). Over all samples, 12 species were Chamaephytes (plants with buds that are 0.1–0.5 M above ground), 38 species were Cryptophytes (plants with below ground dormant tissue), 32 species were Hemicryptophytes (plants with buds that at the ground surface), 34 species were Phanerophytes (trees or shrubs with buds greater than 0.5 m above ground), and 18 species were Therophytes (annuals) (see Appendix A). The frequency distribution of these species by life-form and training intensity is shown in Table 3. Cryptophytes were the most frequent group of species for reference, moderate, heavy, and remediation areas. In contrast, Phanerophytes were the most frequent life-form for light training areas. Therophytes (annuals) were least frequent for reference and light training areas, whereas Chamaephytes were least frequent for moderate and remediation sites. Heavy

Table 2  
Mean and standard deviation (in parentheses) of vegetation characteristics of the different training intensities and remediated plots

| Characteristic              | Training category |                  |                  |                 |                  |
|-----------------------------|-------------------|------------------|------------------|-----------------|------------------|
|                             | Reference         | Light            | Moderate         | Heavy           | Remediation      |
| Understory species richness | 82                | 95               | 78               | 16              | 69               |
| Percent understory cover    | 67.00<br>(10.79)  | 67.87<br>(12.86) | 44.40<br>(18.21) | 4.73<br>(4.50)  | 67.00<br>(17.24) |
| Percent tree cover          | 36.19<br>(4.01)   | 26.10<br>(5.59)  | 0.53<br>(0.92)   | 0.00<br>(0.00)  | 1.92<br>(2.72)   |
| Stand age (years)           | 56.50<br>(24.83)  | 83.67<br>(22.40) | NA <sup>a</sup>  | NA <sup>a</sup> | 7 <sup>b</sup>   |

<sup>a</sup> NA: not applicable because there were no overstory trees in the plots.

<sup>b</sup> Tree age was estimated from planting history.



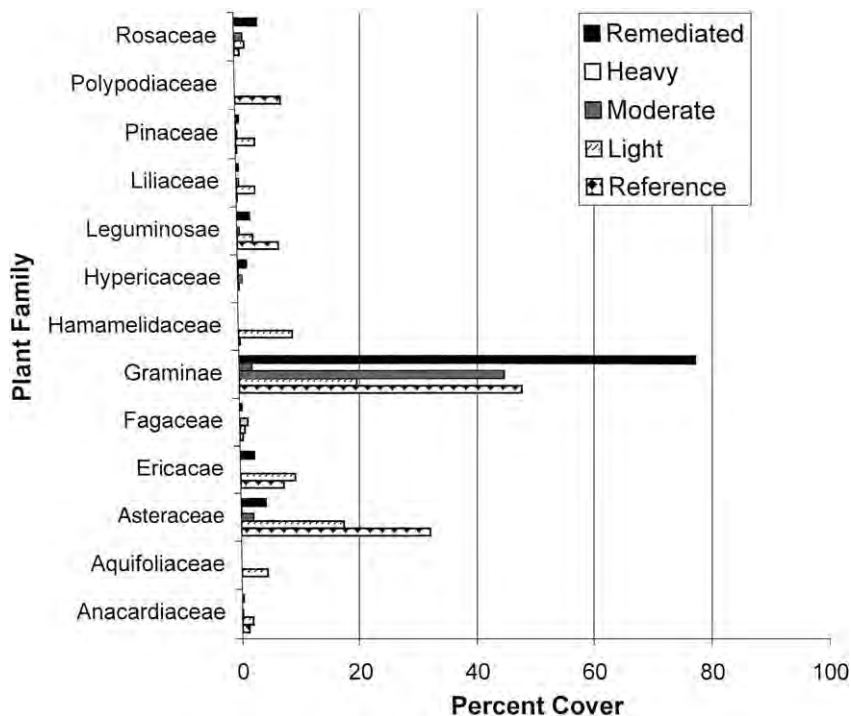


Fig. 3. Percent cover of plant families (for those families with greater than 1% cover) by training category.

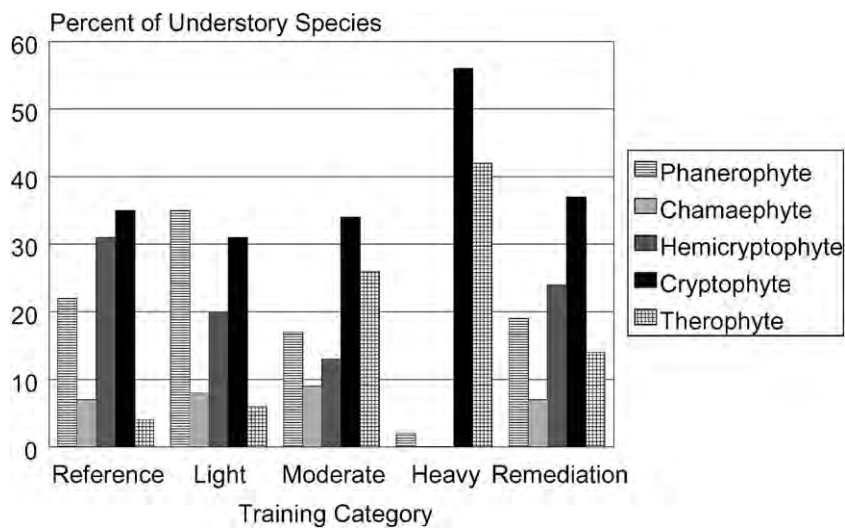


Fig. 4. Life-form distribution by training categories for understory species.



Table 3

Percent of understory species representing major life-forms in training categories (repetitions of species can occur across plots)

| Life-form  | Training category (number of plots) |            |               |            |                  | Total number of times that species of each form were encountered |
|--|-------------------------------------|------------|---------------|------------|------------------|--|
|  | Reference (15)                      | Light (15) | Moderate (15) | Heavy (15) | Remediation (10) |  |
| Phanerophyte   | 22                                  | 35         | 17            | 2          | 19               | 339  |
| Chamaephyte  | 7                                   | 8          | 9             | 0          | 7                | 105  |
| Hemicryptophyte  | 31                                  | 20         | 13            | 0          | 24               | 309  |
| Cryptophyte  | 35                                  | 31         | 34            | 56         | 37               | 490  |
| Therophyte   | 4                                   | 6          | 26            | 42         | 14               | 165  |
| Total number of times that species of each form were encountered | 429                                 | 436        | 268           | 48         | 227              | 1408   |

Table 4

Comparisons of the frequency of understory plants in vegetation cover classes by life-form for five training categories using the Cochran–Mantel–Haenszel statistic (based on rank scores) and single-factor ANOVA

| Statistic                         | Training category (number of plots) |            |               |            |                  |
|-----------------------------------|-------------------------------------|------------|---------------|------------|------------------|
|                                   | Reference (15)                      | Light (15) | Moderate (15) | Heavy (15) | Remediation (10) |
| Cochran–Mantel–Haenszel Statistic | 47.39                               | 57.959     | 76.484        | 75.738     | 18.141           |
| <i>F</i>                          | –                                   | –          | –             | –          | –                |
| <i>P</i>                          | 0.001                               | 0.001      | 0.001         | 0.001      | .0001            |
| ANOVA                             |                                     |            |               |            |                  |
| <i>F</i>                          | 6.75                                | 7.28       | 13.92         | 0.98       | 11.28            |
| <i>P</i>                          | 0.0001                              | 0.0001     | 0.0001        | NS         | 0.0001           |

training sites supported no Chamaephytes or hemicryptophytes.

Differences in the ranks of the cover scores for all life-forms found in the plots (i.e. zeroes were eliminated) was examined using ANOVA and the Cochran–Mantel–Haenszel statistic (Table 4). All training types had fewer species in the higher cover classes than in categories with low cover. There were significant differences in cover ranks by life-forms within reference, light, moderate, and remediation sites, but not within heavy sites. The lack of difference in the heavy training sites likely reflects the paucity of plants found there. The two-way ANOVA revealed significant differences in cover ranks considering both life-form and training intensity (Table 5).

Depth of the soil A horizon, which is used as a measure of impact of military training, differed significantly between categories of training intensity ( $F_{4,65} = 24.3$ ,  $P < 0.001$ ). Light and reference areas had the greatest depth and also the highest variability

Table 5

Two-way analysis of variance of the plant frequency by life-form and training category when reference, light, moderate and heavy training are considered

|                   | Degrees of freedom | <i>F</i> | <i>P</i> |
|-------------------|--------------------|----------|----------|
| Training category | 3                  | 7.50     | .0001    |
| Life-form         | 4                  | 16.15    | .0001    |
| Interaction       | 7                  | 12.44    | .0001    |

(Fig. 5). Depth of the A horizon for heavy, moderate, and remediated sites was consistently small.

## 5. Discussion

Except for distinguishing heavy training areas, these data suggest that neither understory cover nor plant diversity are useful indicators of past training. This inability to discriminate may have occurred because



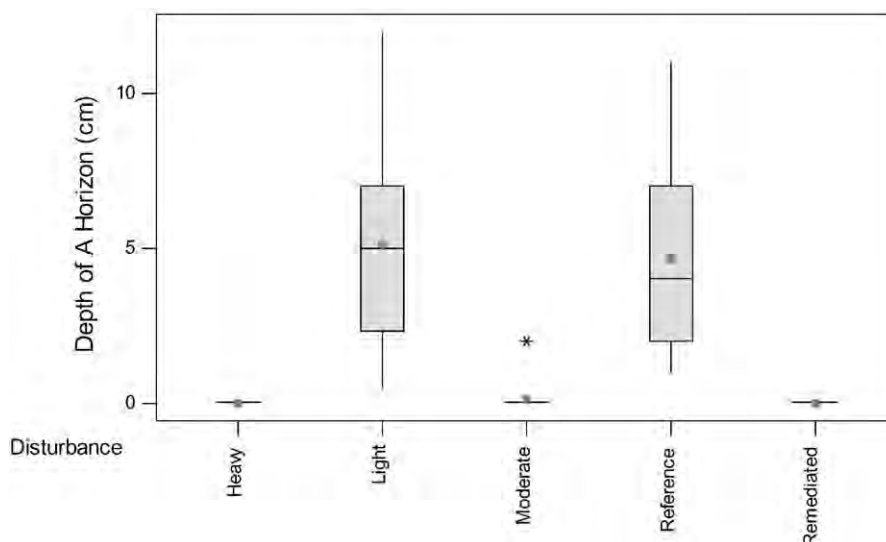


Fig. 5. Boxplot of the depth of the A soil horizon by training intensity type. The bottom and top edges of the box are located at the sample 25th and 75th percentiles. The center horizontal line occurs at the sample median. Means are indicated by solid squares; an outlier value is indicated by an asterisk.

the training areas differed in canopy cover with the light and reference areas being the only ones having significant overstory cover. In those stands, the average age of the trees was 56 years for the reference sites and 83 years for lightly trained sites suggesting it had been at least five–eight decades since a disturbance large enough to induce tree replacement had occurred. However, the influence of canopy cover on understory diversity and cover was not strong. Neither moderate nor remediated sites had an established tree canopy; yet they supported 78 and 69 understory species, respectively, compared to 82 and 95 species for reference and light training areas.

Furthermore, understory cover of remediated areas was equivalent to that of reference and light training sites. Moderate sites averaged 44% understory cover, about two-thirds of that in light, remediated, and reference sites, suggesting that recovery still had to be achieved. Understory species richness and percent cover were quite low for the heavily-used training areas probably because most plants had been removed by repeated tank traffic.

The high diversity and large variation in understory cover of longleaf pine forests and reestablishing vegetation provided a challenge in the use of understory species to distinguish between training impacts

in longleaf pine stands (see [Appendix A](#)). It was not surprising that little bluestem (*Andropogon scoparius*) contributed the highest mean cover over all sites, for it is a characteristic plant of longleaf pine forests ([Dobrowolski et al., 1992](#); [Kirkman et al., 2000](#)). Species that were only identified from one type of training area sometimes were helpful in identifying characteristics of such sites. For example, bracken fern (*Pteridium aquilinum*) was only found in reference sites and is a typical plant of old growth longleaf pine stands. Prickly pear (*Opuntia compressa*) was only found in moderately disturbed sites and can likely withstand the stressful conditions of such sites. The high variability in understory vegetation cover over training categories probably led to the lack of separation by training category by species which required analyses based upon groupings of species into life-forms and families, which are measures of structure and composition (respectively).

In contrast to considering diversity and cover of all species, life-form offered a more effective indicator of past disturbances. Frequency of life-form occurrence distinguishes between military disturbance. Trees and shrubs (Phanerophytes), which may be less affected by foot soldier traffic than other life-forms, dominated cover in light training areas. However, in an extensive



literature review of foot traffic impacts on vegetation, shrubs and trees suffered the longest lasting decrease (Yorks et al., 1997). Our analysis suggested that foot traffic impact on trees and shrubs may not be as intense as on other life-forms. This difference between duration and intensity of disturbance impacts is a necessary distinction (White and Pickett, 1985). Cryptophytes dominate in all other training categories possibly because they are common in the flora due to their ability to withstand ground fires, the natural disturbance of longleaf pine forests. Plants with underground buds are possibly the only vegetation able to withstand heavy tank traffic. In contrast, Therophytes, which are also found in the heavy training areas, likely seeded into sites after mechanized training ceased. Chamaephytes do not contribute more than 1% cover for any training treatment possibly because they are uncommon in the longleaf pine flora and because they are susceptible to all types of traffic.

Previous studies document that life-form reflects impacts following volcanic eruption, grazing, tree thinning, water additions, and soil disturbance (Adams et al., 1987; McIntyre et al., 1995; Stohlgren et al., 1999). In a comparison of treatments designed to reduce hardwood in-growth in longleaf pine forests, fire resulted in the greatest increase in understory species richness and herbaceous groundcover plant densities as compared to herbicide treatments (Provencher et al., 2001). This difference is likely attributed to the fire allowing the survival of plants with their buds below the surface much as dismounted training allows Cryptophytes to survive. Furthermore, life-form changed in the understory after thinning in Douglas fir (*Pseudotsuga menziesii*) plantations (Thomas et al., 1999). Studies from the inner Mongolia Plateau report that life-form is a greater determinant of ecosystem processes than is species richness (Bai et al., 2001).

Plant families are also a useful way to group understory vegetation to reflect differences in training regimes. Of those families that contribute more than one percent cover, light training areas had the highest diversity with 11 families represented whereas heavy training areas had only one family present. Anacardiaceae was the most abundant family in the light training sites (possible because foot soldiers may have avoided poison ivy, one of the common representatives of this family, giving it a competitive advantage over other species that were more readily tramped upon).

Both remediated and reference sites each contained six families with greater than 1% cover, but three of these families were not the same. Ferns (Polypodiaceae) and, in particular, bracken fern (*Pteridium aquilinum*) were distinct to reference sites and can be assumed to be an indicator of the absence of military disturbance.

Graminae was the only family common to all training types. Yorks et al. (1997) report from their literature review of foot traffic impacts that graminoids were found to be most resistant. Grasses contributed very little cover in the heavily trained sites but provided more than 70% cover to the remediated sites. It is not surprising that remediated sites had such high cover of grasses, for recovery efforts of these areas included planting grass seed. The relevant management question is: does such planting bring impacted sites closer to the vegetative characteristic of naturally revegetating sites? We found no family that was distinct to remediated sites. Except for the low percentage of trees and shrubs, life-form distribution of remediated sites is similar to that of both reference and lightly trained sites with Cryptophytes being well represented (Fig. 4). Thus, this analysis suggests that the remediated sites are moving along the pathway toward established vegetation much like that of the reference or lightly trained areas.

Depth of the soil A horizon offers a means independent of observation and vegetative measures to distinguish between the impacts due to military training. The fact the A horizon depth for sites that experienced dismount traffic is not distinct from the reference sites suggests that foot soldier traffic has relatively little impact on the physical conditions of the longleaf pine understory. Yet, the increased percent of trees and shrubs species in the light training areas versus the reference sites cannot be explained by soil properties but is more likely a result of the movement of foot soldiers through the forest.

## 6. Conclusions

We hypothesized that understory diversity and cover sampled from an anthropogenic disturbance gradient within the longleaf pine forests would reveal significant compositional and structural differences that occurred as a result of military training intensity. The confirmation of life-form distribution and plant



family cover as distinguishing features suggests that monitoring programs for longleaf pine forests should include understory vegetation as an ecological indicator. These metrics can serve as surrogate measures of disturbance to the longleaf pine system. Both life-form distribution and plant family cover appear to be useful ways to group the large number of species which occur in the understory of these longleaf pine forests.

Indicators of disturbance that are used for resource management need to be easy to measure, sensitive to stresses, and predictable as to how they respond to stress (Cairns et al., 1993; Stewart and Loar, 1994, Dale and Beyeler, 2001). Selecting indicators for the understory of longleaf pine forests is complicated by the high species diversity. Field classification of understory plants according to life-form and family is relatively straightforward compared to species identification. Both of these attributes are relatively easy and time efficient to measure and interpret. Thus, we recommend that the suite of indicators used for

monitoring longleaf pine ecosystems include these metrics.

## Acknowledgements

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## Appendix A

### Characteristics of understory species found in longleaf pine forests at Fort Benning, GA.

| Botanical name                    | Family           | Raunkiaer life-form<br>(as discussed by <a href="#">Kershaw and Looney, 1985</a> ) |           | Growth-form | Cover |      | Common name                      | Location by impact<br>(R = reference, L = light,<br>M = moderate, H = heavy,<br>D = remediated) |
|-----------------------------------|------------------|--|-----------|-------------|-------|------|----------------------------------|---|
|                                   |                  |  |           |             | Mean  | S.D. |                                  |   |
| <i>Agalinis purpurea</i>          | Scrophulariaceae | Therophyte   |           | Forb        | 0.17  | 0.45 | Gerardia                         | LMD   |
| <i>Albizia julibrissin</i>        | Leguminosae      | Phanerophyte   | Deciduous | Tree        | 0.01  | 0.12 | Silk tree                        | L   |
| <i>Ambrosia artemisiifolia</i>    | Asteraceae       | Therophyte   |           | Forb        | 0.01  | 0.12 | Annual ragweed                   | M   |
| <i>Andropogon scoparius</i>       | Graminae         | Cryptophyte  | Geophyte  | Graminoid   | 2.64  | 2.78 | Little bluestem                  | RLMH  |
| <i>Andropogon ternarius</i>       | Graminae         | Cryptophyte  | Geophyte  | Graminoid   | 1.20  | 2.04 | Splitbeard bluestem              | RLD   |
| <i>Aristida oligantha</i>         | Graminae         | Cryptophyte  | Geophyte  | Graminoid   | 0.86  | 1.72 | Threeawn grass, wire grass       | RLMHD   |
| <i>Aristida purpurascens</i>      | Graminae         | Cryptophyte  | Geophyte  | Graminoid   | 1.27  | 2.07 | Arrowfeather threeawn grass      | RMHD  |
| <i>Aster concolor</i>             | Asteraceae       | Cryptophyte  | Geophyte  | Forb        | 0.26  | 0.63 | Eastern silver aster             | R   |
| <i>Aster dumosus</i>              | Asteraceae       | Cryptophyte  | Geophyte  | Forb        | 0.37  | 0.71 | Rice button aster                | RLD   |
| <i>Aster patens</i>               | Asteraceae       | Cryptophyte  | Geophyte  | Forb        | 0.06  | 0.23 | Spreading aster <sup>a</sup>     | RLMD  |
| <i>Aster tortifolius</i>          | Asteraceae       | Cryptophyte  | Geophyte  | Forb        | 0.77  | 1.49 | White-topped aster               | RLMD  |
| <i>Bulbostylis ciliatifolia</i>   | Cyperaceae       | Therophyte   |           | Forb        | 0.16  | 0.71 | Bulbos rush <sup>a</sup>         | M   |
| <i>Cacalia lanceolata</i>         | Asteraceae       | Hemicryptophyte  | Rosette   | Forb        | 0.04  | 0.20 | Indian-plantain                  | R   |
| <i>Cacalia muhlenbergii</i>       | Asteraceae       | Hemicryptophyte  | Rosette   | Forb        | 0.14  | 0.43 | Great Indian- plantain           | RD  |
| <i>Carya tomentosa</i>            | Juglandaceae     | Phanerophyte   | Deciduous | Tree        | 0.14  | 0.55 | Mockernut hickory                | LD  |
| <i>Cassia nictitans</i>           | Leguminosae      | Therophyte   |           | Forb        | 0.89  | 1.29 | Partridge pea                    | RLMD  |
| <i>Cenchrus longispinus</i>       | Graminae         | Therophyte   |           | Forb        | 0.20  | 0.77 | Sandspurs                        | M   |
| <i>Conyza canadensis</i>          | Asteraceae       | Therophyte   |           | Forb        | 0.46  | 0.85 | Horseweed                        | MHD   |
| <i>Coreopsis major</i>            | Asteraceae       | Cryptophyte  | Geophyte  | Forb        | 0.24  | 0.65 | Greater tickseed                 | R   |
| <i>Crataegus flava</i>            | Rosaceae         | Phanerophyte   | Deciduous | Tree        | 0.47  | 1.10 | Yellow hawthorne                 | RLMD  |
| <i>Crataegus pulcherrima</i>      | Rosaceae         | Phanerophyte   | Deciduous | Tree        | 0.06  | 0.38 | Handsome hawthorne <sup>a</sup>  | LD  |
| <i>Crataegus spathulata</i>       | Rosaceae         | Phanerophyte   | Deciduous | Tree        | 0.01  | 0.12 | Little hop hawthorne             | R   |
| <i>Croton glandulosus</i>         | Euphorbiaceae    | Therophyte   |           | Forb        | 0.06  | 0.23 | Croton                           | MH  |
| <i>Cynodon dactylon</i>           | Graminae         | Cryptophyte  | Geophyte  | Graminoid   | 0.61  | 1.54 | Bermuda grass                    | MH  |
| <i>Desmodium</i> sp.              | Leguminosae      | Cryptophyte  | Geophyte  | Forb        | 0.49  | 0.93 | Ticktrefoil                      | RLM   |
| <i>Desmodium ciliare</i>          | Leguminosae      | Cryptophyte  | Geophyte  | Forb        | 0.24  | 0.60 | Hairy smallleaf ticktrefoil      | RL  |
| <i>Desmodium floridanum</i>       | Leguminosae      | Cryptophyte  | Geophyte  | Forb        | 0.10  | 0.35 | Florida ticktrefoil              | L   |
| <i>Desmodium lineatum</i>         | Leguminosae      | Cryptophyte  | Geophyte  | Forb        | 0.37  | 0.80 | Linear ticktrefoil <sup>a</sup>  | RLM   |
| <i>Desmodium strictum</i>         | Leguminosae      | Cryptophyte  | Geophyte  | Forb        | 0.11  | 0.40 | Pinebarren ticktrefoil           | L   |
| <i>Dichanthelium aciculare</i>    | Graminae         | Cryptophyte  | Geophyte  | Forb        | 0.04  | 0.27 | Needleleaf rosetta grass         | D   |
| <i>Dichanthelium oligosanthos</i> | Graminae         | Cryptophyte  | Geophyte  | Forb        | 0.63  | 1.22 | Bound rosetta grass <sup>a</sup> | RLMD  |



## Appendix A. (Continued)

| Botanical name                  | Family          | Raunkiaer life-form<br>(as discussed by Kershaw<br>and Looney, 1985) |                 | Growth-form | Cover |      | Common name                   | Location by impact<br>(R = reference, L = light,<br>M = moderate, H = heavy,<br>D = remediated) |
|---------------------------------|-----------------|--|-----------------|-------------|-------|------|-------------------------------|---|
|                                 |                 |  |                 |             | Mean  | S.D. |                               |   |
| <i>Dichanthelium sabulorum</i>  | Graminae        | Cryptophyte  | Geophyte        | Forb        | 1.11  | 1.86 | Hemlock rosetta grass         | RLMD  |
| <i>Digitaria violascens</i>     | Graminae        | Cryptophyte  | Geophyte        | Forb        | 0.39  | 1.07 | Crabgrass                     | MHD   |
| <i>Diodea teres</i>             | Graminae        | Therophyte   |                 | Forb        | 0.81  | 1.27 | Poorjoe                       | LMH   |
| <i>Diospyros virginiana</i>     | Ebenaceae       | Phanerophyte   | Deciduous       | Tree        | 1.29  | 1.44 | Common persimmon              | RLMD  |
| <i>Elephantopus tomentosus</i>  | Asteraceae      | Hemicryptophyte  | Partial rosette | Forb        | 0.26  | 0.94 | Devils grandmother            | R   |
| <i>Eragrostis curvula</i>       | Graminae        | Cryptophyte  | Geophyte        | Graminoid   | 0.09  | 0.72 | Weeping lovegrass             | M   |
| <i>Eragrostis capillaris</i>    | Graminae        | Therophyte   |                 | Graminoid   | 1.00  | 2.11 | Slender lovegrass             | LMD   |
| <i>Eragrostis refracta</i>      | Graminae        | Cryptophyte  | Geophyte        | Graminoid   | 0.64  | 1.65 | Coastal lovegrass             | MHD   |
| <i>Erianthus contortus</i>      | Graminae        | Cryptophyte  | Geophyte        | Graminoid   | 0.16  | 0.61 | Sugar cane                    | L   |
| <i>Eupatorium album</i>         | Asteraceae      | Hemicryptophyte  | Proto           | Forb        | 0.41  | 0.89 | White thoroughwort            | RLMD  |
| <i>Eupatorium aromaticum</i>    | Asteraceae      | Hemicryptophyte  | Proto           | Forb        | 0.37  | 1.02 | Aromatic thoroughwort         | RL  |
| <i>Eupatorium capillifolium</i> | Asteraceae      | Hemicryptophyte  | Proto           | Forb        | 0.64  | 1.17 | Small dogfennel               | RLM   |
| <i>Eupatorium coelestinum</i>   | Asteraceae      | Hemicryptophyte  | Proto           | Forb        | 0.03  | 0.17 | Mistflower                    | R   |
| <i>Eupatorium hyssopifolium</i> | Asteraceae      | Hemicryptophyte  | Proto           | Forb        | 0.69  | 1.28 | Hyssopleaf thoroughwort       | RLMD  |
| <i>Euphorbia corollata</i>      | Euphorbiaceae   | Hemicryptophyte  | Proto           | Forb        | 0.16  | 0.40 | Flowering spurge              | RLD   |
| <i>Froelichia gracilis</i>      | Amaranthaceae   | Therophyte   |                 | Forb        | 0.10  | 0.35 | Cottonweed                    | MH  |
| <i>Galium hispidulum</i>        | Rubiaceae       | Chamaephytes   | Active          | Forb        | 0.01  | 0.12 | Coastal beardstraw            | L   |
| <i>Gaylussacia frondosa</i>     | Ericaceae       | Phanerophyte   | Deciduous       | Tree        | 0.06  | 0.29 | Dangleberry                   | R   |
| <i>Gelsemium sempervirens</i>   | Loganiaceae     | Chamaephytes   | Active          | Vine        | 0.07  | 0.26 | Yellow Jasmine                | LM  |
| <i>Gymnopogon ambiguus</i>      | Graminae        | Cryptophyte  | Geophyte        | Grass       | 0.17  | 0.45 | Beard grass                   | RLD   |
| <i>Haplopappus divaricatus</i>  | Asteraceae      | Therophyte   |                 | Forb        | 0.44  | 0.83 | Haplopappus                   | LMHD  |
| <i>Helianthus longifolius</i>   | Asteraceae      | Cryptophyte  | Geophyte        | Forb        | 0.13  | 0.68 | Longleaf sunflower            | D   |
| <i>Heterotheca graminifolia</i> | Asteraceae      | Hemicryptophyte  | Partial Rossett | Forb        | 2.13  | 2.78 | Narrowleaf silkgrass          | RLD   |
| <i>Heterotheca subaxillaris</i> | Asteraceae      | Therophyte   |                 | Forb        | 0.04  | 0.20 | Annual silkgrass <sup>d</sup> | M   |
| <i>Heterotheca gossypina</i>    | Asteraceae      | Hemicryptophyte  | Partial Rosette | Forb        | 0.09  | 0.44 | Hairy silkgrass <sup>a</sup>  | L   |
| <i>Heterotheca mariana</i>      | Composite       | Hemicryptophyte  | Partial Rosette | Forb        | 0.20  | 0.60 | Erect silkgrass <sup>a</sup>  | RL  |
| <i>Hieracium gronovii</i>       | Asteraceae      | Cryptophyte  | Geophyte        | Forb        | 0.03  | 0.17 | Queendevil                    | L   |
| <i>Hypericum gentianoides</i>   | Hypericaceae    | Therophyte   |                 | Forb        | 0.66  | 1.06 | Pineweed                      | LMHD  |
| <i>Hypericum hypericoides</i>   | Hypericaceae    | Phanerophyte   | Deciduous       | Forb        | 0.34  | 0.88 | St. Andrew's cross            | LMD   |
| <i>Ilex glabra</i>              | Aquifoliaceae   | Phanerophyte   | Evergreen       | Shrub       | 0.17  | 1.06 | Inkberry                      | L   |
| <i>Ipomoea pandurata</i>        | Convolvulaceae  | Chamaephytes   | Active          | Vine        | 0.07  | 0.31 | Man of the Earth              | M   |
| <i>Ipomoea purpurea</i>         | Convolvulaceae  | Therophyte   |                 | Forb        | 0.01  | 0.12 | Common morning glory          | D   |
| <i>Lechea minor</i>             | Cistaceae       | Hemicryptophyte  | Proto           | Forb        | 0.17  | 0.48 | Thyme leaf pinweed            | RLMD  |
| <i>Lechea villosa</i>           | Cistaceae       | Hemicryptophyte  | Proto           | Forb        | 0.17  | 0.54 | Hairy pinweed                 | MD  |
| <i>Lespedeza cuneata</i>        | Leguminosae     | Hemicryptophyte  | Proto           | Forb        | 0.24  | 0.77 | Chinese lespodeza             | MD  |
| <i>Lespedeza hirta</i>          | Leguminosae     | Hemicryptophyte  | Proto           | Forb        | 0.37  | 0.76 | Hairy lespodeza               | RLMD  |
| <i>Lespedeza repens</i>         | Leguminosae     | Hemicryptophyte  | Proto           | Forb        | 0.19  | 0.57 | Creeping lespodeza            | MD  |
| <i>Lespedeza stuevei</i>        | Leguminosae     | Hemicryptophyte  | Proto           | Forb        | 0.23  | 0.57 | Tall lespodeza                | RLMD  |
| <i>Liatris elegans</i>          | Asteraceae      | Cryptophyte  | Geophyte        | Forb        | 0.21  | 0.59 | Pinkscale gayfeather          | RL  |
| <i>Liatris graminifolia</i>     | Asteraceae      | Cryptophyte  | Geophyte        | Forb        | 0.17  | 0.45 | Blazing star                  | RL  |
| <i>Liatris tenuifolia</i>       | Asteraceae      | Cryptophyte  | Geophyte        | Forb        | 0.11  | 0.36 | Blazing star                  | RM  |
| <i>Lichens</i>                  |                 |  |                 | Lichens     | 0.61  | 1.69 | Lichen                        | MD  |
| <i>Liquidambar styraciflua</i>  | Hamamelidaceae  | Phanerophyte   | Deciduous       | Tree        | 0.61  | 1.76 | Sweet gum                     | RL  |
| <i>Lithospermum carolinense</i> | Boraginaceae    | Hemicryptophyte  | Proto           | Forb        | 0.01  | 0.12 | Puccoon                       | M   |
| <i>Lobelia puberula</i>         | Campanulaceae   | Hemicryptophyte  | Proto           | Forb        | 0.01  | 0.12 | Downy lobelia                 | L   |
| <i>Mollugo verticillata</i>     | Aizoaceae       | Therophyte   |                 | Forb        | 0.01  | 0.12 | Indian chickweed              | M   |
| <i>moss</i>                     |                 |  |                 | Mosses      | 0.66  | 1.73 | Moss                          | LMD   |
| <i>Muhlenbergia expansa</i>     | Graminae        | Cryptophyte  | Geophyte        | Forb        | 0.11  | 0.55 | Muhly                         | RD  |
| <i>Myrica cerifera</i>          | Myricaceae      | Phanerophyte   | Deciduous       | Shrub       | 0.19  | 0.77 | Southern bayberry             | L   |
| <i>Opuntia compressa</i>        | Cactaceae       | Hemnicryptophyte   | Rosette         | Forb        | 0.03  | 0.17 | Prickly pear                  | M   |
| <i>Paronychia herniarioides</i> | Caryophyllaceae | Therophyte   |                 | Forb        | 0.03  | 0.17 | Coastal Plain nailwort        | M   |
| <i>Paspalum notatum</i>         | Graminae        | Cryptophyte  | Geophyte        | Forb        | 1.33  | 2.50 | Bahiagrass                    | M   |
| <i>Passiflora incarnata</i>     | Passifloraceae  | Hemicryptophyte  | Proto           | Vine        | 0.01  | 0.12 | Maypops                       | M   |
| <i>Petalostemum pinnatum</i>    | Leguminosae     | Hemicryptophyte  | Proto           | Forb        | 0.10  | 0.39 | Summer-farewell               | RL  |
| <i>Pinus echinata</i>           | Pinaceae        | Phanerophyte   | Evergreen       | Tree        | 0.23  | 0.85 | Shortleaf pine                | LH  |
| <i>Pinus palustris</i>          | Pinaceae        | Phanerophyte   | Evergreen       | Tree        | 0.71  | 1.31 | Longleaf pine                 | RLMD  |
| <i>Pinus taeda</i>              | Pinaceae        | Phanerophyte   | Evergreen       | Tree        | 0.14  | 0.62 | Loblolly pine                 | RLD   |
| <i>Polypremum procumbens</i>    | Loganiaceae     | Cryptophyte  | Geophyte        | Forb        | 0.24  | 0.62 | Juniper leaf                  | LMHD  |
| <i>Prunus</i> sp. 1             | Rosaceae        | Phanerophyte   | Deciduous       | Tree        | 0.14  | 0.62 | Plum                          | MD  |
| <i>Prunus serotina</i>          | Rosaceae        | Phanerophyte   | Deciduous       | Tree        | 0.03  | 0.17 | Black cherry                  | LD  |
| <i>Prunus</i> sp. 2             | Rosaceae        | Phanerophyte   | Deciduous       | Tree        | 0.01  | 0.12 | Plum                          | M   |
| <i>Pteridium aquilinum</i>      | Polypodiaceae   | Cryptophyte  | Geophyte        | Forb        | 0.64  | 1.73 | Western brackenfern           | R   |
| <i>Quercus incana</i>           | Fagaceae        | Phanerophyte   | Evergreen       | Tree        | 0.14  | 0.62 | Blue jack oak                 | RLD   |
| <i>Quercus laevis</i>           | Fagaceae        | Phanerophyte   | Deciduous       | Tree        | 0.10  | 0.51 | Turkey oak                    | RL  |
| <i>Quercus laurifolia</i>       | Fagaceae        | Phanerophyte   | Deciduous       | Tree        | 0.10  | 0.42 | Laurel oak                    | RLM   |



## Appendix A. (Continued)

| Botanical name                   | Family           | Raunkiaer life-form<br>(as discussed by Kershaw<br>and Looney, 1985) |                 | Growth-form | Cover |      | Common name                         | Location by impact<br>(R = reference, L = light,<br>M = moderate, H = heavy,<br>D = remediated) |
|----------------------------------|------------------|--|-----------------|-------------|-------|------|-------------------------------------|---|
|                                  |                  |  |                 |             | Mean  | S.D. |                                     |   |
| <i>Quercus margaretta</i>        | Fagaceae         | Phanerophyte   | Deciduous       | Tree        | 0.13  | 0.80 | Shrubby post oak                    | RM  |
| <i>Quercus marilandica</i>       | Fagaceae         | Phanerophyte   | Deciduous       | Tree        | 0.81  | 1.34 | Blackjack oak                       | RL  |
| <i>Quercus nigra</i>             | Fagaceae         | Phanerophyte   | Deciduous       | Tree        | 0.24  | 0.73 | Water oak                           | RLMD  |
| <i>Quercus shumardii</i>         | Fagaceae         | Phanerophyte   | Deciduous       | Tree        | 0.03  | 0.17 | Swamp red oak                       | RM  |
| <i>Quercus stellata</i>          | Fagaceae         | Phanerophyte   | Deciduous       | Tree        | 0.10  | 0.46 | Post oak                            | L   |
| <i>Rhus aromatica</i>            | Anacardiaceae    | Phanerophyte   | Deciduous       | Shrub       | 0.16  | 0.53 | Fragrant sumac                      | MD  |
| <i>Rhus copallina</i>            | Anacardiaceae    | Phanerophyte   | Deciduous       | Shrub       | 1.00  | 1.39 | Flameleaf sumac                     | RLM   |
| <i>Rhynchosia reniformis</i>     | Leguminosae      | Hemicryptophyte  | Proto           | Forb        | 0.50  | 0.76 | Dollar leaf                         | RL  |
| <i>Rubus cuneifolius</i>         | Rosaceae         | Phanerophyte   | Deciduous       | Shrub       | 1.24  | 1.58 | Sand blackberry                     | RLMD  |
| <i>Rubus trivialis</i>           | Rosaceae         | Chamaephytes   | Active          | Shrub       | 0.87  | 1.50 | Dewberry                            | RLMD  |
| <i>Ruellia caroliniensis</i>     | Acanthaceae      | Hemicryptophyte  | Partial rosette | Forb        | 0.03  | 0.17 | Carolina wild petunia               | R   |
| <i>Sassafras albidum</i>         | Lauraceae        | Phanerophyte   | Deciduous       | Tree        | 0.34  | 0.92 | Sassafras                           | RLD   |
| <i>Schrankia microphylla</i>     | Leguminosae      | Phanerophyte   | Deciduous       | Tree        | 0.06  | 0.23 | Sensitive plant                     | RL  |
| <i>Seymeria pectinata</i>        | Scrophulariaceae | Therophyte   |                 | Forb        | 0.10  | 0.49 | Piedmont blacksenna                 | L   |
| <i>Smilax bona-nox</i>           | Liliaceae        | Chamaephytes   | Active          | Forb        | 0.20  | 0.91 | Saw greenbriar                      | RLM   |
| <i>Smilax glauca</i>             | Liliaceae        | Chamaephytes   | Active          | Forb        | 0.26  | 0.53 | Cat greenbriar                      | RLMD  |
| <i>Smilax laurifolia</i>         | Liliaceae        | Chamaephytes   | Active          | Forb        | 0.14  | 0.49 | Laurel greenbriar                   | LMD   |
| <i>Smilax rotundifolia</i>       | Liliaceae        | Chamaephytes   | Active          | Forb        | 0.14  | 0.49 | Roundleaf greenbriar                | LM  |
| <i>Solanum carolinense</i>       | Solanaceae       | Hemicryptophyte  | Proto           | Forb        | 0.21  | 0.45 | Carolina horsenettle                | RLMD  |
| <i>Solidago leavenworthii</i>    | Asteraceae       | Cryptophyte  | Geophyte        | Forb        | 0.01  | 0.12 | Leavenworth goldenrod <sup>a</sup>  | M   |
| <i>Solidago odora</i>            | Asteraceae       | Cryptophyte  | Geophyte        | Forb        | 0.91  | 1.38 | Anise-scented goldenrod             | RLD   |
| <i>Solidago stricta</i>          | Asteraceae       | Cryptophyte  | Geophyte        | Forb        | 0.33  | 0.77 | Slender goldenrod <sup>a</sup>      | RLD   |
| <i>Solidago tenuifolia</i>       | Asteraceae       | Cryptophyte  | Geophyte        | Forb        | 0.16  | 0.58 | Narrow goldenrod <sup>a</sup>       | RL  |
| <i>Stipulicida setacea</i>       | Caryophyllaceae  | Hemicryptophyte  | Rosette         | Forb        | 0.03  | 0.17 | Stipulicida <sup>a</sup>            | M   |
| <i>Strophostyles umbellata</i>   | Leguminosae      | Chamaephytes   | Active          | Vine        | 0.20  | 0.77 | Trailing strophostyles <sup>a</sup> | R   |
| <i>Stylosanthes biflora</i>      | Leguminosae      | Hemicryptophyte  | Proto           | Forb        | 0.16  | 0.50 | Pencil flower                       | RL  |
| <i>Tephrosia virginiana</i>      | Leguminosae      | Chamaephytes   | Suffrutescent   | Forb        | 0.59  | 1.34 | goat's rue                          | RMD   |
| <i>Toxicodendron quercifolia</i> | Anacardiaceae    | Chamaephytes   | Suffrutescent   | Vine        | 0.50  | 1.06 | poison ivy                          | RL  |
| <i>Trichostema dichotomum</i>    | Lamiaceae        | Therophyte   |                 | Forb        | 0.01  | 0.12 | Blue curls                          | L   |
| <i>Triplasis americana</i>       | Graminae         | Cryptophyte  | Geophyte        | Grass       | 0.17  | 0.64 | Sand grass                          | RLM   |
| <i>Triplasis purpurea</i>        | Graminae         | Cryptophyte  | Geophyte        | Grass       | 0.46  | 1.14 | Purple sand grass                   | LMHD  |
| <i>Vaccinium arboreum</i>        | Ericaceae        | Phanerophyte   | Deciduous       | Shrub       | 0.74  | 1.68 | Farkleberry                         | RLD   |
| <i>Vaccinium eliotii</i>         | Ericaceae        | Phanerophyte   | Deciduous       | Shrub       | 0.60  | 1.42 | Elliots blueberry                   | RL  |
| <i>Vaccinium myrsinites</i>      | Ericaceae        | Phanerophyte   | Evergreen       | Shrub       | 0.64  | 1.50 | Shiny blueberry                     | RLD   |
| <i>Vaccinium stamineum</i>       | Ericaceae        | Phanerophyte   | Deciduous       | Shrub       | 1.06  | 1.80 | Deerberry                           | RLD   |
| <i>Vernonia angustifolia</i>     | Asteraceae       | Hemicryptophyte  | Proto           | Forb        | 0.26  | 0.61 | Tall ironweed                       | RL  |
| <i>Viola palmata</i>             | Violaceae        | Hemicryptophyte  | Proto           | Forb        | 0.06  | 0.29 | Palmed violet                       | R   |
| <i>Vitis rotundifolia</i>        | Vitaceae         | Chamaephytes   | Passive         | Vine        | 0.04  | 0.20 | Muscadine                           | L   |
| <i>Wahlenbergia marginata</i>    | Campanulaceae    | Hemicryptophyte  | Proto           | Forb        | 0.01  | 0.12 | Southern rockbell                   | D   |
| <i>Yucca filamentosa</i>         | Liliaceae        | Hemicryptophyte  |                 | Shrub       | 0.19  | 0.43 | Adams needle                        | LMD   |

<sup>a</sup> No common name was provided by taxonomy books; so, these common names were derived from the Latin name or description.

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# Soil microbial biomass and community composition along an anthropogenic disturbance gradient within a long-leaf pine habitat

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## Abstract

Some of the finest surviving natural habitat in the United States is on military reservations where land has been protected from development. However, responsibilities of military training often require disturbance of that habitat. Herein, we show how the soil microbial community of a long-leaf pine ecosystem at Fort Benning, Georgia responds to military traffic disturbances. Using the soil microbial biomass and community composition as ecological indicators, reproducible changes showed increasing traffic disturbance decreases soil viable biomass, biomarkers for microeukaryotes and Gram-negative bacteria, while increasing the proportions of aerobic Gram-positive bacterial and Actinomycete biomarkers. Soil samples were obtained from four levels of military traffic (reference, light, moderate, and heavy) with an additional set of samples taken from previously damaged areas that were remediated via planting of trees and ground cover. Utilizing 17 phospholipid fatty acid (PLFA) variables that differed significantly with land usage, a linear discriminant analysis with cross-validation classified the four groups. Wilks' lambda for the model was 0.032 ( $P < 0.001$ ). Overall, the correct classifications of profiles was 66% (compared to the chance that 25% would be correctly classified). Using this model, 10 observations taken from the remediated transects were classified. One observation was classified as a reference, three as light trafficked, and six as moderately trafficked. Non-linear artificial neural network (ANN) discriminant analysis was performed using the biomass estimates and all of the 61 PLFA variables. The resulting optimal ANN included five hidden nodes and resulted in an  $r^2$  of 0.97. The prediction rate of profiles for this model was again 66%, and the 10 observations taken from the remediated transects were classified with four as reference (not impacted), two as moderate, and four as heavily trafficked. Although the ANN included more comprehensive data, it classified eight of the 10 remediated transects at the usage extremes (reference or heavy traffic). Inspection of the novelty indexes from the prediction outputs showed that the input vectors from the remediated transects were very different from the data used to train the ANN. This difference suggests as a soil is remediated it does not escalate through states of succession in the same way as it descends following disturbance. We propose to explore this hysteresis between disturbance and recovery process as a predictor of the resilience of the microbial community to repeated disturbance/recovery cycles. © 2001 Elsevier Science Ltd. All rights reserved.

**Keywords:** Phospholipid fatty acid (PLFA); Habitat disturbance; Recovery; Artificial neural network (ANN)

## 1. Introduction

Managers at military installations provide land for training of military personnel. Often, such activities are inconsistent with sustainable land use practices.

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An effective monitoring program capable of predicting recovery from training use is essential to ensuring the long-term viability of these lands. Previously, the majority of these monitoring programs have focused on biodiversity of terrestrial macroorganisms that have a long period of recovery and require expensive monitoring specialists. Microorganisms that can be quantitatively monitored with chemical biomarkers have been largely overlooked despite their complete integration and dependency with the macroworld (Zak et al., 1994; Lee, 1991).

Microbial biomass in soil has a turnover time of less than a year and can react quickly to conditions of nutrients, moisture, temperature and the type and amount of soil organic matter levels (Paul, 1984). Viable microbial biomass is integral for nutrient storage and cycling (Rice et al., 1996), soil aggregate formation (Tisdall and Oades, 1982), and other ecological factors such as filtering, buffering, and gene reserves (Blum, 1998). Soil microbial biomass and community composition have been shown to be sensitive indicators of changes in nutrient type (Peacock et al., 2001; Kirchner et al., 1993), botanical composition (Borga et al., 1994), pollutant toxicity (Stephen et al., 1999), and climate change (Zogg et al., 1997). Because the microbial community integrates the physical and chemical aspects of the soil and responds to anthropogenic activities, it can be considered a biological indicator of soil quality (Rice et al., 1996). The microbial viable biomass, community composition, and nutritional status can be readily measured by analysis of extracted lipids to provide rational endpoints for many disturbance/recovery processes (White et al., 1998).

In this study, we investigated the utility of using the soil microbial biomass and community composition as ecological indicators of change along an anthropogenic disturbance gradient. The disturbance gradient included the duration and type of traffic in military training areas in a long-leaf pine habitat. The hypothesis was that duration and intensity of disturbance (traffic) in the long-leaf pine ecosystem would be reflected in changes in the soil microbial community biomass and structure. These changes could be quantitatively measured by phospholipid ester-linked fatty acid analysis (PLFA). In addition, we used two different data analysis techniques to classify disturbance, the first, a linear discriminant analysis classified transects based on 17 PLFA variables, the second, a non-linear

artificial neural network analysis which included all 61 PLFA variables and the biomass in which to base predictions. Herein, we compare these two analyses.

## 2. Materials and methods

### 2.1. Study site

This study was conducted at the Fort Benning Army Installation located in the lower Piedmont Region and lower coastal plane of central Georgia and Alabama, near Columbus, Georgia. The post consists of approximately 73,650 ha of river valley terraces and rolling terrain. The climate at Fort Benning is humid and mild with rainfall occurring regularly throughout the year. Annual precipitation averages 105 cm with October being the driest month. The majority of the soils at the installation are heavily weathered Ultisols.

This study encompassed training areas that have been subjected to a range of military traffic. Disturbance of the soil ecosystem due to training includes the direct removal or damage of terrestrial vegetation, digging activities, dislocation, and compaction from vehicles, erosion, and sedimentation. The degree and extent of the impacts of training activities within a compartment are dependent upon the type of activity, number of personnel training, and how frequently the compartment is exposed to activity. Furthermore, training activity typically occurs irregularly throughout a compartment, creating localized gradients of disturbance within individual compartments.

### 2.2. Soil sampling

Soil cores were collected in the Autumn of 1999. To avoid cross contamination in between each sample, the soil cores were washed in solvent (methanol) and sterile distilled water. Cores were approximately 20 cm in length and 2 cm in width. For each core taken, the depth of the core and the presence/absence of an "A" horizon was reported. Five samples were taken from separate plots at each transect (14 transects  $\times$  5 = 70 samples, Table 1). Of the transects selected, three were reference transects (with stand ages 28, 68, and 74 years); three were heavy usage (undergoing tracked vehicle training); three were moderate usage (areas adjacent to tracked vehicle training); three were



Table 1  
Sample design

| Transect | Number of samples | Disturbance |
|----------|-------------------|-------------|
| A        | 5                 | Reference   |
| E        | 5                 | Reference   |
| M        | 5                 | Reference   |
| D        | 5                 | Light       |
| L        | 5                 | Light       |
| N        | 5                 | Light       |
| C        | 5                 | Moderate    |
| I        | 5                 | Moderate    |
| K        | 5                 | Moderate    |
| B        | 5                 | Heavy       |
| H        | 5                 | Heavy       |
| J        | 5                 | Heavy       |
| F        | 5                 | Remediated  |
| G        | 5                 | Remediated  |

light usage (infantry training), and two came from a site currently undergoing remediation (previous heavy disturbance, currently trees and groundcover planted and no usage). Samples were stored at  $-80^{\circ}\text{C}$  prior to analysis.

### 2.3. PLFA analysis

PLFA analysis was performed using previously reported precautions (White and Ringelberg, 1998). Soil samples (5 g) were extracted with the single-phase chloroform–methanol buffer system of Bligh and Dyer, 1954, as modified (White et al., 1979). The total lipid extract was fractionated into neutral lipids, glycolipids, and polar lipids by silicic acid column chromatography (Guckert et al., 1985). All results presented in this paper are for the polar lipid fraction. The polar lipids were transesterified to the fatty acid methyl esters (FAMES) by a mild alkaline methanolysis (Guckert et al., 1985).

The FAMES were analyzed by capillary gas chromatography with flame ionization detection on a Hewlett-Packard 5890 series 2 chromatograph with a 50 m non-polar column (0.2 mm i.d., 0.11  $\mu\text{m}$  film thickness). Preliminary peak identification was performed by comparison of retention times with known standards. Definitive identification of peaks was accomplished by gas chromatography/mass spectroscopy of selected samples using a Hewlett-Packard 6890 series gas chromatograph interfaced to a

Hewlett-Packard 5973 mass selective detector using a 20 m non-polar column (0.1 mm i.d., 0.1  $\mu\text{m}$  film thickness).

Fatty acids are named according to the convention  $X:Y\omega Z$ , where “X” stands for the number of carbon atoms in the chain, “Y” for the number of unsaturations, and “Z” the number of carbon atoms from the terminal methyl end of the molecule to the first unsaturation encountered. Prefixes; “i”=*iso*-branched, “a”=*anteiso*-branched, “10me”=*methyl branch on the 10th carbon from the carboxylate end*, “Br”=*branched at unknown location*, and “Cy”=*cyclopropyl*. The suffixes “c” and “t” stand for the *cis* and *trans*, geometric isomers of the unsaturation, respectively. When different fatty acids had the same designation, they were distinguished by lower case letters suffixes a, b, etc. (Gunstone and Herslöf, 1992).

### 2.4. Statistical analysis

Biomass (pmol/g PLFA) and relative proportion (mol%) of specific PLFA were used to test the null hypothesis that degree of land disturbance would not influence the composition of the soil microbial communities. To test that hypothesis, an analysis of variance (ANOVA) using the General Linear Model STATISTICA procedure (Statsoft Inc., Tulsa, OK) for a completely randomized design with five treatments was used. The values reported are least square means of 15 replicates, except in the case of the remediated treatment which contained 10 replicates (total  $n = 70$ ). Standard errors of the means were determined. Differences in the mean proportions of PLFA in each treatment were tested using Tukey’s Honest-Significant-Difference procedure. A hierarchical cluster analysis (Ward’s method, 1-Pearson  $r$ ) was used to discover how the PLFAs that differed significantly with treatment were clustered.

A linear discriminant analysis with cross-validation (SAS Institute Cary, NC) was chosen to classify the observations into one of the four usage classes ( $n = 60$ , 15 observations in each group) based on the degree of land disturbance. Only those PLFA that comprised at least 1% of any profile were included in the analysis. Therefore, fatty acids that may have been unreliably quantified were not included. Before statistical analysis, arcsine square root transformation was applied



to the mol% PLFA data. After truncation, a one-way ANOVA was conducted on the remaining PLFAs, and those that differed significantly with usage were included in the model.

### 2.5. Artificial neural network analysis

Neural network identification was performed with early stopping by cross-validation and topology optimization by bootstrapping (selection criteria; median cross-validated error), using microcortex web based neural computing environment ([www.microcortex.com](http://www.microcortex.com)).

## 3. Results

Degree of land use resulted in a significant difference in the microbial biomass estimates (PLFA), for the highly trafficked soil ( $P < 0.05$ , Fig. 1). If it is assumed that 1 pmol of PLFA is equivalent to  $2.5 \times 10^4$  bacterial cells (Balkwill et al., 1988; Pinkart et al., 2000), then bacterial density in the soils ranged from approximately  $7.7 \times 10^8$  cells  $g^{-1}$  in the reference soil to  $3.8 \times 10^7$  cells  $g^{-1}$  in the heavily trafficked soil.

The soil currently undergoing restoration contained an average of  $5.8 \times 10^8$  cells  $g^{-1}$ .

PLFA analysis identified 61 fatty acids all of which are commonly found in soil environments (Peacock et al., 2001). Of the 61 fatty acids detected and quantified, 28 were highly significant according to a one-way ANOVA ( $P < 0.001$ ). Mean separations were conducted on the 28 PLFAs using Tukey's Honest-Significant-Difference procedure and the results are presented in Table 2. Generally, the short-chain normal saturated PLFA (14:0, 15:0, and 16:0) decreased with increasing traffic, while the longer chain normal saturated PLFA (18:0 and 20:0) increased with increasing traffic. Monounsaturated and polyunsaturated PLFAs decreased with increasing traffic, whereas the methyl-branched saturated PLFAs increased with increasing traffic. An exploratory hierarchical cluster analysis (Ward's method, 1-Pearson  $r$ ) was conducted using the 28 highly significant variables (Fig. 2). Two primary clusters emerged. The first contained predominantly short-chain saturated, monounsaturated, and polyunsaturated PLFA, while the second contained long-chain saturates, methyl-branched monounsaturated, and saturated PLFA. A secondary cluster derived from the first

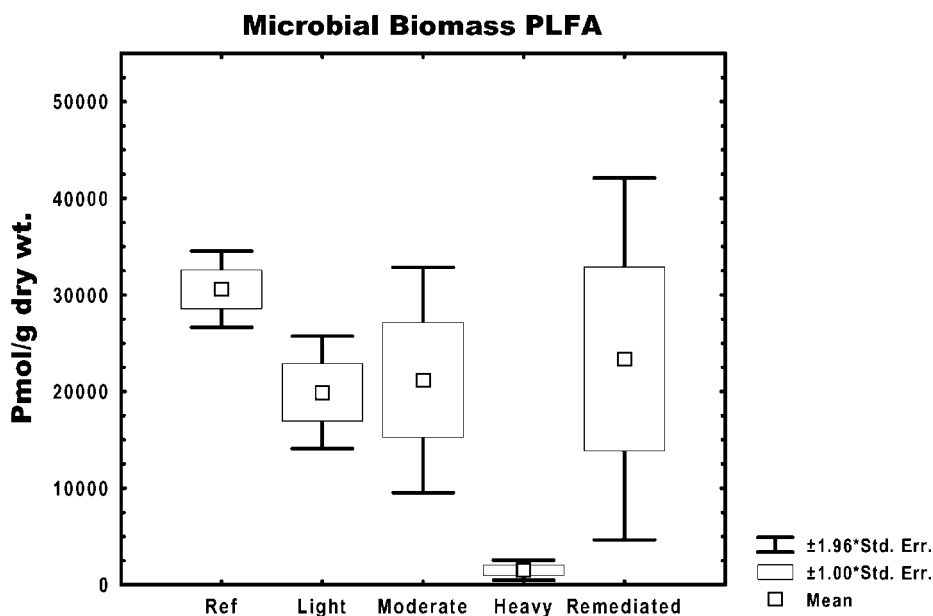


Fig. 1. Microbial biomass PLFA of samples from the four disturbance categories and samples undergoing remediation. The reference samples contain the highest microbial biomass followed by light and moderate disturbance and finally heavy disturbance.



Table 2  
Mean relative proportions of PLFAs by treatment

| PLFA                                | Reference | Light   | Moderate | Heavy    | Remediated |
|-------------------------------------|-----------|---------|----------|----------|------------|
| <b>General</b>                      |           |         |          |          |            |
| 14:0                                | 0.53 a    | 0.50 a  | 0.46 a   | 0.18 b   | 0.40 a     |
| 15:0                                | 0.65 a    | 0.57 a  | 0.58 a   | 0.26 b   | 0.51 a     |
| 16:0                                | 12.8 a    | 12.81 a | 13.31 a  | 10.08 b  | 13.31 a    |
| 18:0                                | 3.04 c    | 3.47 c  | 4.20 b   | 5.12 a   | 3.42 c     |
| 20:0                                | 0.96 b    | 0.81 b  | 1.06 b   | 1.63 a   | 0.76 b     |
| <b>Gram-negative bacteria</b>       |           |         |          |          |            |
| 15:1                                | 0.08 a    | 0.09 a  | 0.09 a   | 0.02 b   | 0.06 ab    |
| 16:1w7 c                            | 2.79 a    | 2.57 ac | 2.29 bc  | 1.99 b   | 2.73 ac    |
| 16:1w5 c                            | 1.9 a     | 1.82 a  | 1.96 a   | 1.52 a   | 2.49 b     |
| 17:1                                | 0.16 ab   | 0.18 a  | 0.10 b   | 0.06 c   | 0.11 abc   |
| 18:1w5 c                            | 0.89 a    | 0.96 a  | 0.89 a   | 0.35 b   | 0.95 a     |
| Cy19:0                              | 12.84 a   | 13.53 a | 10.33 b  | 11.34 ab | 8.92 b     |
| <b>Eukaryote (plant and fungal)</b> |           |         |          |          |            |
| 18:2w6                              | 5.74 a    | 5.90 a  | 3.63 b   | 1.00 c   | 6.51 a     |
| 18:1w9 c                            | 8.49 a    | 8.32 a  | 7.79 a   | 6.08 b   | 8.71 a     |
| 20:3w3                              | 0.08 a    | 0.09 a  | 0.07 ab  | 0.01 b   | 0.08 a     |
| 20sat                               | 2.08 a    | 2.08 a  | 1.32 b   | 0.00 b   | 0.01 b     |
| poly20 a                            | 0.13 a    | 0.16 a  | 0.03 b   | 0.02 b   | 0.02 b     |
| poly20 b                            | 0.18 ab   | 0.34 a  | 0.28 a   | 0.07 b   | 0.35 a     |
| <b>Actinomycetes type</b>           |           |         |          |          |            |
| i14:0                               | 0.19 a    | 0.13 a  | 0.15 a   | 0.03 b   | 0.19 a     |
| br16:0 a                            | 0.80 b    | 1.06 b  | 1.23 b   | 3.89 a   | 0.92 b     |
| br16:0 b                            | 0.16 a    | 0.12 a  | 0.07 ab  | 0.01 b   | 0.07 ab    |
| i16:0                               | 3.22 ab   | 2.84 b  | 3.86 a   | 3.36 ab  | 3.51 ab    |
| i17:1w7 c                           | 1.44 b    | 1.49 ab | 1.82 a   | 1.73 ab  | 1.6 ab     |
| 10me16:0                            | 3.87 b    | 3.96 b  | 4.46 ab  | 4.82 a   | 4.13 ab    |
| i17:0                               | 2.17 c    | 2.24 c  | 3.76 b   | 4.79 a   | 2.95 c     |
| a17:0                               | 2.14 b    | 2.10 b  | 2.70 a   | 2.96 a   | 2.67 a     |
| 17:0                                | 0.64 c    | 0.71 bc | 0.76 b   | 0.88 a   | 0.67 bc    |
| i10me16:0                           | 1.26 c    | 1.34 c  | 3.35 b   | 6.04 a   | 1.93 c     |
| 12me18:0                            | 0.68 c    | 0.66 c  | 1.45 b   | 2.43 a   | 1.48 b     |

Treatments followed by the same letter are not significant at  $\alpha = 0.05$ .

primary cluster contained short-chain normal saturated and 16 carbon monounsaturates. The remaining secondary clusters contained mostly 18–20 carbon mono- and polyunsaturates. Secondary clusters derived from the second primary cluster included long-chain normal saturates and methyl-branched fatty acids.

A linear discriminant analysis with cross-validation was chosen to classify the observations into one of four classes ( $n = 60$ , 15 observations in each group) based on the degree of land disturbance. The first task was to reduce the number of variables to be included in the model. Only those PLFA that comprised at least 1% of any profile were included in the analysis;

Table 3  
PLFAs included in the discriminant model

|           |           |          |
|-----------|-----------|----------|
| a15:0     | i17:0     | 18:1w9 c |
| i1:0      | a17:0     | 18:0     |
| 16:1w7 c  | Cy17:0    | 10me18:0 |
| br16:0 a  | 17:0      | Cy19:0   |
| i17:1w7 c | i10me16:0 | 20sat    |
| 10me16:0  | 18:2w6    |          |

therefore, fatty acids that may have been unreliably quantified were not included. Before statistical analysis arcsine square root transformation was applied to the mol% PLFA data. After this truncation, a one-way ANOVA was conducted on the remaining PLFAs, and those that differed significantly with usage were included in the model. The resulting model included 17 descriptor variables (Table 3). Wilks' lambda for the model was 0.32 ( $P < 0.01$ ). Overall, the error estimates for the model were 33% and the generalized distance between groups is reported in Table 4. Only the first four treatments were used to construct the model. Once the model was complete, the 10 observations taken from the remediated transects were classified. One observation was classified as a reference, three as lightly trafficked, and six as moderately trafficked.

A non-linear artificial neural network discriminant analysis (ANN) was performed using the biomass estimates and all of the 61 PLFA variables. The resulting ANN included five hidden nodes and resulted in an  $r^2$  of 0.97. The correct classification of profiles for this model was 66%, and six of the PLFAs had sensitivity values above 3%. As with the linear discriminant model, once the ANN model was complete, it was used to classify the observations from the remediated transects. Four of the observations were classified as reference, two as moderate, and four as heavily trafficked.

#### 4. Discussion

The four categories of traffic in this study varied in the amount and diversity of the floristic component (Dale and Beyeler, 2001). In addition, soil carbon and nitrogen concentrations and stocks as well as the carbon to nitrogen ratios, differed significantly with degree of traffic (Garten et al., 2001). Soil compaction due to the amount of traffic was also significantly



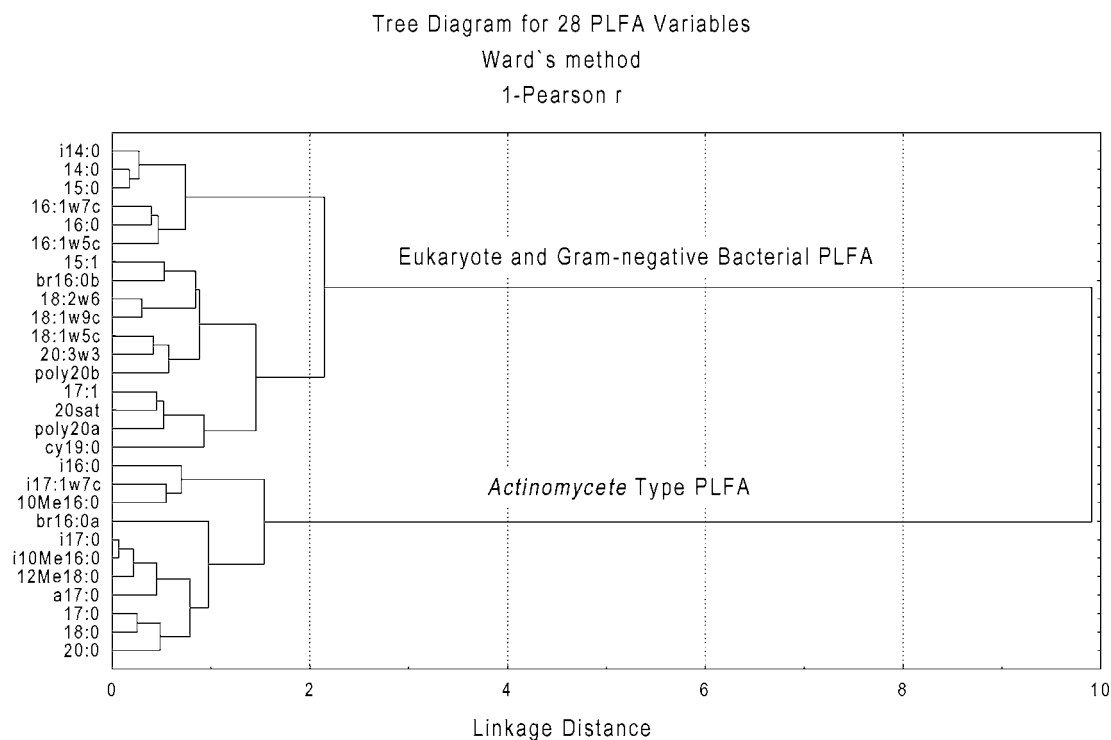


Fig. 2. Cluster analysis of significant PLFA variables (mol%). Two primary clusters emerged, the first contained primarily PLFAs indicative of eukaryote microorganisms (polyunsaturates) and Gram-negative bacteria (monounsaturates). While the second contained PLFA indicative of the *Actinomycetes* (methyl-branched saturates).

different along the disturbance gradient (Garten et al., 2001).

Myers et al., 2001 states, “microbial metabolism in soil is limited by the availability and types of organic

substrates, and therefore it is plausible that ecosystems which differ floristically will produce litter with chemically distinct substrates that will differentially foster microbial growth.” Soil microbial community

Table 4  
Number of observations and percent classified into usage

|                              | Reference  | Light     | Moderate   | Heavy      | Total    |
|------------------------------|------------|-----------|------------|------------|----------|
| Reference (%)                | 11 (73.33) | 4 (26.67) | 0 (0)      | 0 (0)      | 15 (100) |
| Light (%)                    | 6 (40)     | 8 (53.33) | 1 (6.67)   | 0 (0)      | 15 (100) |
| Moderate                     | 0 (0)      | 2 (13.33) | 11 (73.33) | 2 (13.33)  | 15 (100) |
| Heavy                        | 0 (0)      | 1 (6.67)  | 4 (26.67)  | 10 (66.67) | 15 (100) |
| Error count estimates        |            |           |            |            |          |
| Rate (%)                     | 26.6       | 46.6      | 26.6       | 33.3       | 33.3     |
| Priors (%)                   | 25         | 25        | 25         | 25         |          |
| Generalized squared distance |            |           |            |            |          |
| Reference                    | 0          | 5.65      | 40.47      | 77.8       |          |
| Light                        | 5.65       | 0         | 23.22      | 52.95      |          |
| Moderate                     | 40.47      | 23.22     | 0          | 11.85      |          |
| Heavy                        | 77.8       | 52.95     | 11.85      | 0          |          |



composition and biomass differed along the gradient as measured by the PLFA analysis. Biomass content in these soils decreased with increasing traffic and was significantly lower in highly trafficked soil (Fig. 1). Specific PLFA components can be related to certain subsets of the microbial community, and PLFA patterns can be used to define changes in the community composition. Using the ANOVA results (Table 2), the reference and the lightly trafficked soil contained on average more PLFAs indicative of eukaryotes (including plant associated PLFAs) and Gram-negative bacteria (Wilkinson, 1988), whilst the more trafficked soils contained relatively more PLFAs associated with Actinomycetes (O'Leary and Wilkinson, 1988; Verma and Khuller, 1983). The HCL analysis (Fig. 2) using variable clustering, illustrates this point. Over the disturbance gradient, when PLFA markers for eukaryotes and Gram-negative bacteria were high the PLFAs indicative of the Actinomycetes were low. Monounsaturated PLFAs are indicative of predominantly Gram-negative bacteria (White et al., 1996). An increase in the amount and type of carbon sources has been shown to increase monounsaturated PLFAs (Peacock et al., 2001; Bossio and Scow, 1998; Macnaughton et al., 1999). The loss of monounsaturated PLFAs with traffic indicates a loss of these types of bacteria. Terminally branched saturated PLFA in aerobic environments are indicative of Gram-positive bacteria, including *Arthrobacter* and *Bacillus* spp. (White et al., 1996). Many of these types of bacteria can be spore formers and can exist in environments that are lower in overall organic carbon content and higher metabolic refractiveness (Boylan and Ensign, 1970; Keynan and Sandler, 1983). Mid-chain branched saturated PLFA are primarily indicative of Actinomycete type bacteria in surface soils. It has been stated that since these bacteria grow conidia, they are able to better survive in relatively harsh soil environments (desiccation and heat). This may give these bacteria a competitive advantage in the heavily trafficked areas (Alexander, 1998). Polyunsaturated PLFA, shows significant decreases due to traffic and indicates the loss of fungi and microbial grazers that follows the loss of bacterial microorganisms.

Analysis of the soil microbial community PLFA in a predictive linear discriminant model was successful in distinguishing the amount of traffic a soil received. Inspecting the generalized squared distance results

from the linear discriminant analysis revealed that the reference and lightly trafficked soils were very close in terms of the microbial community composition (Table 4). In comparison, the moderate and heavily trafficked soils were very different. Indeed, when observations were classified during model validation, most of the misclassifications were between the reference and lightly trafficked soils.

To more fully explore the relationships between the soil disturbance and the microbial community composition, without assumptions of normal distributions or linear relationships, a non-linear artificial neural network discriminant model was applied to the data. The overall predictive effectiveness for correct profile classification for the model was 66%, which was the same for the linear discriminant model. However, the ANN was constructed and optimized using all of the 61 PLFAs and included the biomass parameter. As with the linear analysis, most of the misclassifications occurred between traffic categories that were close (i.e. moderate being classified as heavy). However, when the ANN was used to predict the status of the remediated transects, eight of the 10 samples were classified as either reference or heavy traffic. Inspection of the novelty indexes from the prediction outputs showed that the input vectors from the remediated transects were very different from the data used to train the ANN. This result is not surprising, as when the soil is remediated, it does not escalate through states of succession in the same way it descended by disturbance. In other words, in this case there is not a sliding scale on which the ecosystem recovery can be measured, but a new community succession is taken, initiated by the remediation efforts (planting of groundcover and trees). We propose to further explore this hysteresis between disturbance and recovery process in the microbiota by coupling the lipid biomarker analysis to the specific analysis of community DNA. DNA analyzed by PCR of rDNA with separation of components comprising >1% of the community by denaturing gradient gel electrophoresis (DGGE) for sequencing and identification adds more specificity to the quantitative analysis of the lipids. This provides a much clearer model of disturbance and recovery from pollution than either analysis done separately (Stephen et al., 1999).

The subtlety of the hysteresis between disturbance and recovery was not detected with the linear discriminant model that showed no bias toward extreme



classifications. With the linear discriminant analysis, most samples undergoing remediation were classified as either moderate or light usage with one sample being classified as reference. Since this analysis was linear and only used 17 descriptor variables, the resultant predictions may be of a more general nature, whereas the ANN used the complete matrix in which to base predictions. Regardless, the predictions of the linear analysis could be accepted and used to aid stakeholders in management of the land use.

## 5. Conclusions

The goal of this project was to explore the possibility of using the soil microbial community as an ecological indicator signaling the degree of environmental degradation along a disturbance gradient. The analysis based on the soil PLFA was successful, reflected above-ground changes, and provided an index of the degree of land disturbance (traffic) the soil received. Both linear discriminant and non-linear ANN analysis were able to adequately classify the degree of disturbance. However, there were drawbacks when the ANN and linear discriminant models were used to predict stages of soil recovery in remediated transects. The linear discriminant model was shown to be a fairly robust but perhaps coarse measure of remediation efforts, and the ANN was sufficiently sensitive to detect subtleties in recovery not detected with the linear discriminant analysis, but in current form could not be relied on to classify remediated samples. The inclusion of data reflecting remediation in these models could possibly make them capable of monitoring a more complete process of soil degradation and recovery.

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**Experimental response of understory plants to mechanized disturbance in longleaf pine forests**

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**Running Head**

Understory indicators of disturbance

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## Abstract

Fort Benning, Georgia, contains active infantry training grounds and more than 65,000 ha of soils suitable for longleaf pine (*Pinus palustris*) forest, a greatly reduced forest type in North America. As longleaf pine forests are the primary habitat for the federally endangered red-cockaded woodpecker (*Picoides borealis*), land managers at this installation must both maintain conditions for mechanized training and conserve this forest habitat. Understory vegetation serves a central role in longleaf pine forests and a recent study suggests that impacts from military activity are best indicated by grouping understory species into Raunkiaer life-forms. This study tests this hypothesis by experimentally manipulating a longleaf pine forest using a mechanized vehicle. In May 2003, a D7 bulldozer removed extant vegetation and surface soil organic matter along three treatment transects. Braun-Blanquet vegetation surveys were conducted within plots in mid and late summer through 2003 and 2004. Repeated measures analyses of variance were utilized to determine the effectiveness of total plant cover, bare ground cover, litter cover, species richness, family richness, and life-form cover in indicating the response of understory vegetation to mechanized disturbance. Total understory cover, species richness, and family richness did not show a treatment effect; however, bare ground cover was greater in treatment transects while litter cover was greater in control transects. Both bare ground and litter covers converged toward control transect values. While these results suggest that the understory largely recovered from mechanized disturbance after two growing seasons, results by life-form revealed a functional shift in the vegetation. Phanerophytes (trees and shrubs) in treatment transects exhibited reduced cover and did not converge toward control transect values. In contrast, Therophytes (annuals), Hemicryptophytes (plants with buds at ground surface), and Cryptophytes (plants with underground buds) displayed an interaction between time and treatment suggesting that, after an initial reduction, mechanized disturbance may eventually increase their cover relative to controls. These findings support the hypothesis that plant functional types, such as Raunkiaer life-forms, best indicate the response of understory vegetation to mechanized disturbance in longleaf pine forests. Raunkiaer life-forms provide a consistent measure of understory structure that can be assessed across varying plant communities.

## Key Words

*Ecological indicator; Pinus palustris; Picoides borealis; plant functional types; Raunkiaer life-form; repeated-measures analysis of variance; tracked-vehicle disturbance*

## Introduction

Characterizing the response of ecosystems to disturbance constitutes a central and challenging principle of an ecological approach to land management (Dale et al. 2000). Disturbance, whether anthropogenic or natural, disrupts the structure, composition, or function of ecosystems. In order to monitor the effects of disturbance, land managers need certain metrics, or ecological indicators, that express changes in structure, composition, or function of ecosystems (Dale and Beyeler 2001, Niemi and McDonald 2004). Thus, ecological indicators serve as surrogates that attempt to simplify and synthesize underlying ecosystem dynamics (Murtaugh 1996). Ideally, ecological indicators should be sensitive, readily measurable, consistent in their response, integrative across ecosystem components and broad environmental gradients, and anticipatory of changes in ecological condition that can be averted by management action (Noss 1990, Dale and Beyeler 2001).



The application of ecological indicators has typically focused on particular species within an ecosystem, often in regard to the conservation status of species of concern (Kremen 1992, Niemi and McDonald 2004). Ecological indicators based on particular species are limited spatially to the range of that species and may not be widely applicable across large areas. However, ecological indicators may also be selected from measures of ecosystem composition, structure, or function (Noss 1990). Ecological indicators based on these components of ecosystems have the potential to provide a consistent metric of ecological condition for land management across a range of environmental gradients and geographical extents.

Longleaf pine (*Pinus palustris*) forests were abundant in presettlement forests of southeastern North America extending across approximately 37 million ha from Texas to Virginia; however, their range has been reduced to approximately 1 million ha currently (Frost 1993). These forests are a fire-adapted system featuring an open canopy of pine or mixed pine-hardwood trees, and a diverse understory often dominated by grasses and forbs (Frost 1993, Landers et al. 1995). Longleaf pine's tolerance to frequent (2 to 8 years) understory fire enables its dominance in these forests as it can quickly grow from a protected seedling stage to that of a 1 or 2 m sapling within a few years (Outcalt 2000). In contrast, hardwoods are found more often as sprouts in the understory as both hardwoods and other pine species have greater fire susceptibility (Outcalt 2000). Without periodic fire, hardwoods and other pines are released from the understory eventually replacing longleaf pine (Landers et al. 1995). This change impacts the red cockaded woodpecker (*Picoides borealis*), a federally endangered species that is adapted to longleaf pine forest habitat (James et al. 2001).

Fort Benning, Georgia, contains greater than 65,000 ha of soils capable of supporting longleaf pine (Dale et al. 2002). Although wiregrass is the most common understory plant in longleaf pine forests east of Mississippi (Peet 1993), it is not typical of the communities at Fort Benning (Dale et al. 2002). Instead, bracken fern (*Pteridium aquilinum*) is more characteristic of these understory communities as they transition between sandhills and xeric upland piedmont longleaf woodlands (Golden 1979, Peet and Allard 1993, Dilustro 2002). The uniqueness of the vegetation at Fort Benning frames an ecological question that is not unique: What is the best approach to monitoring the response of a forest community to ongoing human land use?

At Fort Benning, land managers for the military require indicators of ecological condition in response to military training that are not only applicable across this installation but potentially also across other installations in the southeast. This requirement reflects a need to elucidate indicators that are general enough to be applied across a geographical range, but specific enough to provide detailed information on ecological condition.

Dale et al. (2002) conducted an observational study of understory vegetation at Fort Benning across a military training gradient including: reference areas with little or no training, light impact areas with dismounted and individual training use, moderate impact areas that receive some tracked-vehicle activity, and heavy impact areas that receive substantial tracked-vehicle activity. The study sought to identify indicators of understory vegetation as the understory influences the subsequent dynamics of longleaf pine forests. Dale et al. (2002) found that typical metrics of understory vegetation, including total cover of vegetation, species richness, and indicator species were not useful in characterizing the intensity of military disturbance. Total cover and species richness did not distinguish between training classes and insufficient species were found across all training classes to discern a particular indicator species. Instead, Dale et al. (2002) concluded that plant functional types, specifically Raunkiaer life-forms, best indicated the response of understory vegetation to military training.



Raunkiaer's (1934) classification system delineates life-forms by the height of perennating buds (see also Kershaw and Looney 1985). While Raunkiaer envisioned life-forms as a functional response of plants to unfavorable climate, his approach has since been also successfully applied to the functional response of plants to disturbance (McIntyre et al. 1995, Rusch et al. 2003). This study presents an experimental test of the efficacy of Raunkiaer life-forms as indicators of military disturbance to understory vegetation. Specifically, this study asks whether Raunkiaer life-forms serve as a useful indicator of tracked-vehicle disturbance when compared with more standard metrics of understory vegetation within a training compartment at Fort Benning.

## Methods

### *Experimental design*

This study was set within longleaf pine habitat in the K-11 training compartment at Fort Benning, Georgia. In May 2003, soil disturbance was generated through the movement of a D7 bulldozer with its blade lowered to create three, 50 m treatment transects. This treatment not only approximated the movement of mechanized vehicles often utilized in training activities at Fort Benning, but also constituted a more uncommon and severe disturbance than the more frequent fire-induced disturbances in longleaf pine ecosystems. The bulldozer removed most vegetation and organic matter from the soil surface exposing a sandy substrate. A paired control transect was established 5 m adjacent to each disturbance treatment transect. Although the K11 compartment has historically experienced little military training, the three paired transects were located within an area previously manipulated by fire and thinning treatments in 2002, replicating typical management prescriptions at Fort Benning. Along each transect, which were approximately parallel with elevation contour lines, 10 circular plots were chosen at random distances. Each plot encompassed 0.568 m<sup>2</sup>. Transects, not plots, were considered independent replicates to minimize the effects of pseudoreplication on statistical inference (Hurlbert 1984). Of the 60 subplots across three replicate transects, six plots were not considered for analysis because of inconsistent treatment application. Four control plots were impacted by the tracks of the D7 bulldozer as it traveled between transects, one treatment plot was not impacted by the lowered blade of the bulldozer, and one control plot encompassed a portion of a stream and was consistently impacted by the presence of water.

### *Vegetation Sampling*

Within each plot all vegetation less than 1 m in height was surveyed using a modified form of the Braun-Blanquet (1932) cover system (Clarke 1986, Dale et al. 2002) (Table 1). Surveys were conducted a total of four times over two growing seasons: June 2003, September 2003, June 2004, and October 2004. Cover classes were assigned to each species present, total plant cover, bare ground, and leaf litter.

Species identification relied on Radford et al. (1968), Godfrey (1988), and Weakley (2004). Species identifications were confirmed with voucher specimens at the University of Tennessee Herbarium and species nomenclature follows the Integrated Taxonomic Information System (U.S. Department of Agriculture 2004a). Consistent with Dale et al. (2002), functional classifications for all vascular taxa were delimited using Raunkiaer's (1934) life-form classification system. Classification of species into life-forms was assisted by life-history information from Radford et al. (1968), The National PLANTS Database (U.S. Department of Agriculture, Natural Resources Conservation Service 2004), and Fire Effects Information System



(U.S. Department of Agriculture 2004b). Following McIntyre et al. (1995) and Dale et al. (2002), plants were classified as Phanerophytes (trees or shrubs with buds greater than 20 to 30 cm above the ground surface), Chamaephytes (vines with buds near the ground surface), Hemicryptophytes (plants with buds at the ground surface), Cryptophytes (plants with perennating tissue at a depth of at least 2 to 3 cm), and Therophytes (annual plants). Cover values were assigned for each life-form by plot.

While the timing of the field surveys in the mid- and late-growing season facilitated plant identification, not all vegetation could be identified to the species level. In these cases, identification terminated with an assessment of either taxonomic family or genus. As species were aggregated into both taxonomic families and life-forms, this limitation generally did not prevent the inclusion of specimens from the analysis. *Dichanthelium* species were distinguished from the *Panicum* genus by the presence of overwintering rosettes (Weakley 2004), which also defined their respective functional classification as Hemicryptophytes. *Panicum virgatum* was the only species identified in its genus and was classified as a Cryptophyte. Accordingly, all other *Panicum* individuals not identifiable to species were also considered Cryptophytes. By mid-summer, *Pteridium aquilinum* often exhibited senesced fronds; however, these fronds were not readily distinguishable from those of prior growing seasons and were considered as leaf litter.

#### Data analysis

Data were analyzed using Matlab (Mathworks, 2004). Cover-abundance values of total plant, litter, bare ground, and Raunkiaer life-forms were converted to percentage midpoints and summed by life-form per plot. An arcsine square root transformation for proportional data was employed to approximate a normal distribution (Zar 1996). Species and family richness counts were log<sub>10</sub> transformed. Cover and count values for treatment and control replicates (n=6) used the average of the plot values within each transect. Repeated measures analysis of variance (ANOVAR) was employed to compare values between treatment and control transects (df = 1,  $\alpha$  = 0.05) and to analyze values through time and the interaction of time with treatment (df = 3,  $\alpha$  = 0.05). ANOVAR has been widely applied to the analysis of ecological trends through time (see Gurevitch and Chester 1986, Moser et al. 1990, Potvin et al. 1990, Meredith and Stehman 1991, Nuzzo et al. 1996). Orthogonal polynomials were used to decompose the effect of time and its interaction with disturbance treatment into linear, quadratic and cubic trends (df = 1,  $\alpha$  = 0.05) with the assumption that sampling points were distributed evenly during the growing seasons (Keppel 1991). While responses are typically characterized using the more parsimonious linear and quadratic trends, cubic trends were also used because of their potential to express seasonal variations in understory metrics over the four surveys in two growing seasons. The assumption of sphericity in the RMANOVA was evaluated using Mauchly's test and corrected if necessary using the more conservative significance level from either the Greenhouse-Geisser or Huynh-Feldt tests (Potvin et al. 1990). All means and standard errors were re-expressed into original units in figures.

#### Results

102 species from 40 families were identified across all treatment and control plots over two growing seasons (Appendix). Vegetation in the mechanized-vehicle disturbance treatment transects exhibited significant differences from those in the control transects across all metrics of understory vegetation except for Chamaephyte cover; however, the form of these differences (i.e.



whether arising from treatment effects or interactions with treatment through time) varied by metric (Tables 2 and 3). Plot metrics (Table 2 and Figure 1) showed a significant, mean treatment effect only for bare ground and litter cover. Throughout the two growing seasons, bare ground was greater in the treatment transects while litter cover was reduced. The influence of time had a significant mean effect as well as a significant interaction with disturbance treatment on all plot metrics.

The effect of time and its interaction with disturbance treatment is further evident in the significance of most linear orthogonal contrasts for the plot metrics. Total plant cover increased linearly through time across control and treatment transects, but treatment transects increased linearly their cover at a greater rate than control transects. Conversely, bare ground cover decreased linearly across transects, but this reduction was greater for treatment transects. Litter cover increased linearly through time across transects with a greater increase observed in treatment transects, although this interaction was not expressed in any of the component contrasts. Species richness and family richness responded similarly with treatment transects increasingly linearly at a greater rate than the control transects. Total cover, species richness, and family richness all also showed significant quadratic trends through time; however, quadratic contrasts only expressed the overall effect of time more strongly than linear contrasts for species and family richness. The greater increases of treatment transect values for total cover and family richness were also better represented by quadratic contrasts. Cubic trends were observed only for total plant cover but this contrast was better explained by a linear trend.

Understory cover by life-form (Table 3 and Figure 2) exhibited a marginally significant disturbance treatment effect for Phanerophytes over two growing seasons. The observed power of the ANOVAR for Phanerophytes with a significance level of 0.05 was 0.523. Relaxing the significance level to 0.10 (see Stevens 2002) increased the observed power to 0.706, leading to a substantial reduction in Type II error and suggesting that the treatment effect is significant for Phanerophytes. Phanerophytes and Chamaephytes both displayed a time effect, with an increasing linear trend, but no interaction of treatment with time. Phanerophytes also showed a significant quadratic interaction of treatment with time, but since the overall interaction term was not significant, this result may be spurious. Hemicryptophytes, Cryptophytes, and Therophytes showed a significant effect of time and interaction of time with treatment. This interaction was expressed linearly for these three life-forms, but only Hemicryptophytes displayed a linear effect of time. Hemicryptophytes, Cryptophytes, and Therophytes also showed a quadratic trend through time, but this trend was better expressed by a linear contrast for Hemicryptophytes.

## Discussion

The significant effects of either disturbance treatment or its interaction with time shown by each of these ten understory vegetation metrics, excepting Chamaephytes, demonstrate that most of these metrics provide useful information in characterizing the response of understory vegetation to mechanized disturbance. The challenge from a land management perspective is to identify those metrics that are most ideal for indicating the response of understory vegetation. Additionally, all ten metrics also showed a significant effect of time averaged over control and treatment transects. These responses highlight the dynamic character of the longleaf pine understory as it responded to prescribed burning and selective thinning in this watershed during 2002. These management actions, which replicate the historical disturbance regime, are used to restore longleaf pine habitat at Fort Benning as forests receive understory burns approximately every three years and selective thinning every nine years (Dilustro 2002).



Although total plant cover in treatment transects was initially reduced by mechanized disturbance, its greater rate of increase, shown by the interaction terms, resulted in a return to control transect values by the end of the first growing season. Thus, averaged over two years, total plant cover was not sensitive to mechanized disturbance. Likewise, species and family richness also returned to control transect values after one growing season. The similarity in mean values of species and family richness reflects the taxonomic diversity of the understory, which generally resulted in the presence of species from different families within a plot.

In contrast to those three plot metrics, bare ground and litter covers were both sensitive to mechanized disturbance over these two years. Additionally, both bare ground and litter covers converged on their respective control plot covers. Bare ground cover in treatment transects decreased more quickly than control transects while litter cover increased more quickly than control transects. Although all five of these plot metrics are readily measurable and potentially applicable across environmental gradients and understory communities, bare ground and litter covers show greater sensitivity to mechanized disturbance. The relatively transient sensitivity of total plant cover, species richness, and family richness would require monitoring surveys within months of a mechanized training activity and would not capture the longer recovery of bare ground and litter covers.

Relying on either bare ground or litter covers as indicators of ecological condition would suggest that the understory had largely recovered from the effects of mechanized disturbance after two years. However, Raunkiaer life-forms provide additional insight into the functional response of the vegetation that contradicts this interpretation. Phanerophytes, the most prevalent life-form cover in the understory, not only were sensitive to the disturbance treatment after two years, but also exhibited no trend of converging on cover values in control transects. Chamaephytes were not sensitive to mechanized disturbance and showed little interaction of disturbance and time. Hemicryptophytes, Cryptophytes, and Therophytes were also not disturbance sensitive, but did show an interaction with treatment as their cover values returned to control transect values after the first growing season. Additionally, the trend of these three lifeforms suggest that their cover may promoted by mechanized disturbance after two growing seasons.

Overall, the responses of these life-forms demonstrate that mechanized disturbance changes the relative structure of the understory vegetation. Phanerophytes, with buds exposed above the ground surface, are more susceptible to mechanized disturbance than the other lifeforms. These changes persist for at least two years and are not reflected in plot metrics of total plant, bare ground, or litter covers. The treatment effects of these statistical inferences highlight the importance of the temporal scale in this study. Had this study been conducted for only one growing season, it is likely that more metrics would have expressed a treatment effect. In contrast, if this study was continued for additional growing seasons, possibly only Phanerophytes would continue to demonstrate a treatment effect.

While this experiment only examined the response of understory vegetation within one training compartment at Fort Benning, the observational findings of Dale et al. (2002) provide evidence that tracked vehicles induce similar responses across this installation. Specifically, Dale et al. (2002) found that Phanerophyte, Chamaephyte, and Hemicryptophyte covers decreased in areas of heavy tracked-vehicle training while Cryptophyte and Therophyte covers increased. In areas of moderate tracked-vehicle training, Chamaephyte and Cryptophyte covers did not vary substantially from reference values, suggesting that this experimental treatment was more similar to a moderate tracked-vehicle training intensity. Furthermore, this experiment was



only able to reproduce the intensity (sensu White and Pickett 1985), not the duration of repeated tracked-vehicle activity. This difference in experimental treatment may explain why Therophyte cover did not increase and Hemicryptophyte cover did not decrease in the K11 experiment.

The response and recovery of vegetation to tracked-vehicle disturbance has also been investigated across a range of environments within North America. Common among these studies is a general trend from perennial to annual plant species and a decrease in woody vegetation (Demarais et al. 1999, Milchunas et al. 2000). For example, Lathrop (1983) observed reduce cover and density of perennial shrubs 36 years after General Patton's armored maneuvers in the Mojave Desert. At Fort Hood Military Reservation in Texas, Johnson (1982) reported a shift from perennial to annual herbs and a reduction in density and cover of woody understory plants following disturbance by tracked vehicles. Tracked-vehicle disturbance reduced the cover of perennial grasses and sagebrush (*Artemisia tridentata*) at the Idaho Army National Guard Orchard Training Area (Watts 1998). At the Pinon Canyon Maneuver Site in Colorado tracked vehicle maneuvers enabled the establishment of annual cool-season grasses and reduced the presence of perennial warm-season grasses as well as the densities of trees, shrubs, and succulents (Shaw and Diersing 1990, Milchunas et al. 1999). Although these general plant functional types may not be directly analogous to Raunkiaer life-forms, they do suggest that similar functional responses may occur in a variety of ecosystems. This optimism is tempered somewhat by the observation that some plant functional type responses are dependent upon plant community (e.g. Milchunas et al. 2000); however, these patterns may reflect longer-term competitive relationships between plant functional types.

Increased bare ground and decreased litter covers have also been reported as indicators of tracked-vehicle disturbance (e.g. Wilson 1988, Ayers 1994, Shaw and Diersing 1990, Watts 1998, Prosser et al. 2000, Milchunas et al. 2000). These two metrics are well-suited for indicating the impact of tracked-vehicle disturbance, particularly bare ground cover as it relates the absence of vegetation, but the results of this experiment show that they have less value than life-forms for indicating the recovery of understory vegetation structure in a longleaf pine forest.

The distinction between bare ground and litter covers and life-form covers in characterizing the impact and response of tracked-vehicle disturbance underlies an important consideration in utilizing ecological indicators. Plant functional types, such as Raunkiaer life-forms, should be considered as part of a suite of ecological indicators describing vegetation condition (Dale et al. 2004). While each ecosystem is likely to have its own particular species of concern or species that may be serve as indicators within a community type, plant functional types can provide a general metric that may be comparable across community types. For instance, although Dale et al. (2005) reported an increase in *Polypremum procumbens* during the first growing season following this experimental disturbance, the applicability of this species as an indicator will be limited for community types within its geographical range. For instance, the Department of Defense alone is responsible for ecosystem management on 10 million hectares within the United States (Goodman 1996) and almost half of this land, 4.8 million hectares, is the responsibility of the U.S. Army (Shaw and Diersing 1990). These lands span a variety of plant communities and restrict a species-based approach for comparing the responses of vegetation communities to land-use impacts. Finally, plant functional types are also responsive to a range of ecological disturbance types, both natural and anthropogenic (McIntyre et al. 1999), and accordingly have the potential to serve as ecological indicators under a variety of circumstances.



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## APPENDIX

A list of species identified and corresponding Raunkiaer life-forms assigned is provided in  
ESA's Electronic Data Archive: *Ecological Archives* XXXX-XXX-XX.



1 Table 1: Key of cover-abundance classes modified from the Braun-Blanquet (1932) system.

| Cover-abundance<br>class | Species cover and distribution<br>characteristics | Percentage<br>Midpoints |
|--------------------------|---|-------------------------|
| 0                        | No plants present                                 | 0                       |
| 1                        | Less than 1% cover; 1-5 small individuals         | 0.25                    |
| 2                        | Less than 1% cover; many small individuals        | 0.5                     |
| 3                        | Less than 1% cover; few large individuals         | 0.75                    |
| 4                        | 1-5% cover  | 3.0                     |
| 5                        | 5-12% cover                                       | 8.5                     |
| 6                        | 12-25% cover                                      | 18.5                    |
| 7                        | 25-50% cover                                      | 37.5                    |
| 8                        | 50-75% cover                                      | 62.5                    |
| 9                        | 75-100% cover                                     | 87.5                    |

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| Measure / effects | Between plots | Within plots | WITHIN PLOT CONTRASTS |            |         |
|-------------------|---------------|--------------|-----------------------|------------|---------|
|                   |               |              | Linear                | Quadratic  | Cubic   |
| Total cover       | 2.308         |              |                       |            |         |
| Time              |               | 97.202***    | 152.516***            | 110.827*** | 12.059* |
| Time × plot       |               | 18.739***    | 24.854**              | 56.407***  | 4.188   |
| Bare ground cover | 205.347***    |              |                       |            |         |
| Time              |               | 17.544***    | 25.197**              | 2.068      | 0.132   |
| Time × plot       |               | 9.471**      | 13.308*               | 1.859      | 0.016   |
| Litter cover      | 36.229**      |              |                       |            |         |
| Time              |               | 10.970***    | 12.902*               | 1.369      | 0.436   |
| Time × plot       |               | 3.582*       | 3.948                 | 0.327      | 2.872   |
| Species richness  | 0.337         |              |                       |            |         |
| Time              |               | 19.128***    | 19.375*               | 33.846**   | 0.711   |
| Time × plot       |               | 15.314***    | 25.607**              | 4.158      | 0.186   |
| Family richness   | 0.140         |              |                       |            |         |
| Time              |               | 7.465**      | 4.415                 | 85.209***  | 0.236   |
| Time × plot       |               | 9.799**      | 9.716*                | 39.507**   | 1.866   |

1 \*  $P \leq 0.05$ , \*\*  $P \leq 0.01$ , \*\*\*  $P \leq 0.001$

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3 Table 2.  $F$  statistics from repeated measures analyses of variance (mixed two-factor within-subjects design) and orthogonal contrasts

4 that summarize responses of understory vegetation, leaf litter, and bare ground to mechanized vehicle disturbance at Fort Benning,

5 Georgia. Cover values are arcsine square root transformed and count values are log-transformed.

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| Measure / effects       | Between plots      | Within plots | WITHIN PLOT CONTRASTS |           |       |
|-------------------------|--------------------|--------------|-----------------------|-----------|-------|
|                         |                    |              | Linear                | Quadratic | Cubic |
| Phanerophyte<br>Time    | 7.074 <sup>†</sup> | 13.061***    | 15.823*               | 3.412     | 1.396 |
| Time × plot             |                    | 0.317        | 0.182                 | 8.670*    | 0.248 |
| Chamaephyte<br>Time     | 0.009              | 7.960**      | 22.755**              | 3.656     | 0.815 |
| Time × plot             |                    | 1.089        | 4.085                 | 0.134     | 0.362 |
| Hemicryptophyte<br>Time | 0.001              | 26.159***    | 43.080**              | 11.596*   | 6.642 |
| Time × plot             |                    | 9.640**      | 20.362*               | 0.001     | 0.419 |
| Cryptophyte<br>Time     | 0.019              | 4.263*       | 5.930                 | 8.179*    | 0.080 |
| Time × plot             |                    | 4.843*       | 8.025*                | 0.398     | 2.905 |
| Therophyte<br>Time      | 2.120              | 5.578*       | 0.527                 | 9.012*    | 5.962 |
| Time × plot             |                    | 3.492*       | 20.045*               | 2.388     | 0.056 |

1   <sup>†</sup>  $P = 0.056$ , \*  $P \leq 0.05$ , \*\*  $P \leq 0.01$ , \*\*\*  $P \leq 0.001$

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3   Table 3.  $F$  statistics from repeated measures analyses of variance (mixed two-factor within-subjects design) and orthogonal contrasts

4   that summarize responses of understory vegetation cover by Raunkiaer life form to mechanized vehicle disturbance at Fort Benning,

5   Georgia. Cover values are arcsine square root transformed.

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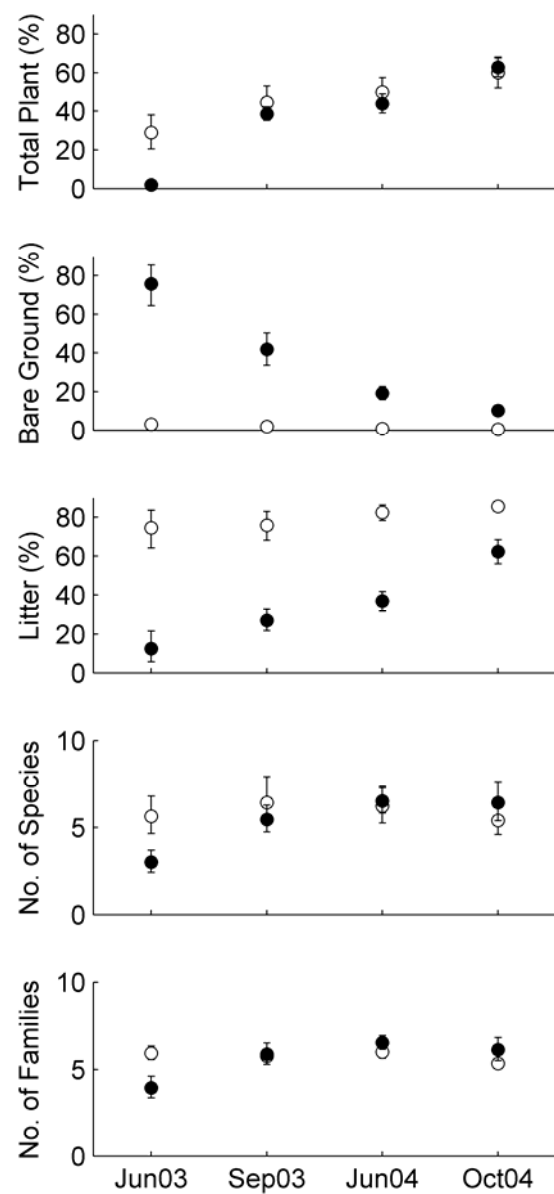


**Figures**

Fig. 1. Response of understory plants following mechanized disturbance in May, 2003 characterized by (A) total cover, (B) visible bare ground, (C) litter cover, (D) species richness, and (E) family richness. Data are transformed means  $\pm$  1 SE re-expressed into original units.

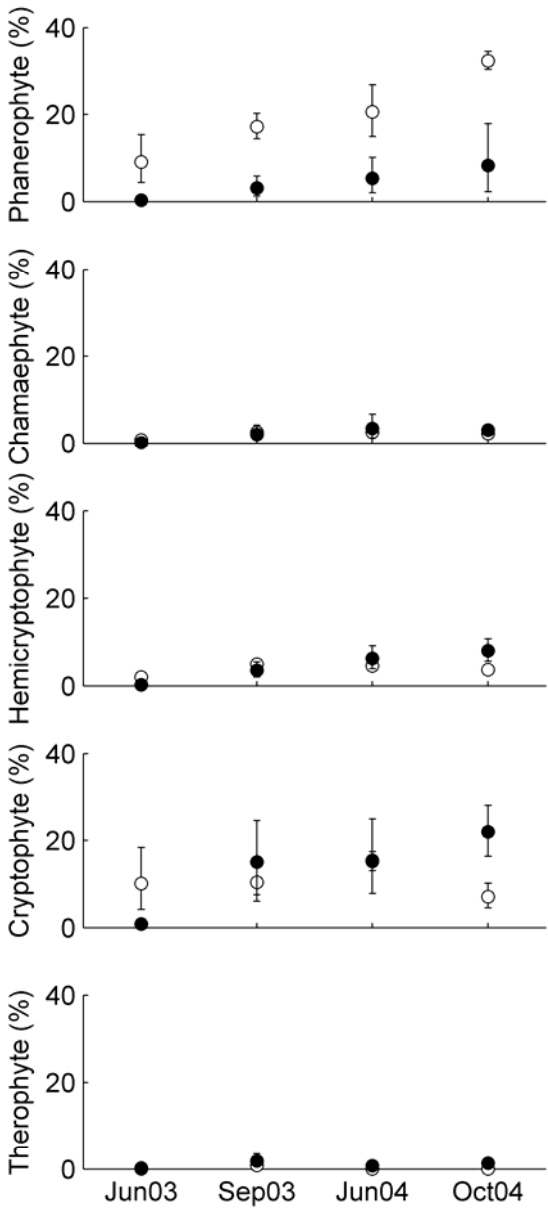
Fig. 2. Response of understory plants following mechanized disturbance in May, 2003 characterized by Raunkiaer life-form cover from: (A) phanerophytes, (B) chamaephytes, (C) hemicryptophytes, (D) cryptophytes, and (E) therophytes. Data are transformed means  $\pm$  1 SE re-expressed into original units.





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Contributions of soil, microbial, and plant indicators to land management of Georgia pine  
forests

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ABSTRACT

Characterizing how resource use and management activities affect ecological conditions is necessary to document and understand ecological changes. Resource managers on military installations have the delicate task of balancing the need to train soldiers effectively with the need to maintain ecological integrity. Ecological indicators can play an important role in the management process by providing feedback on the impacts that training has on environmental characteristics, including soil chemistry, soil microbes, and vegetation. A discriminant function analysis was conducted to determine whether ecological indicators could differentiate between different levels of military use. A combination of ten indicators explained 90% of the variation among plots from five different military-use levels. Results indicated that an appropriate suite of ecological indicators for military resource managers includes vegetation, microbial, and soil



1 characteristics. Since many of the indicators are correlated, managers will have freedom  
2 to choose indicators that are relatively easy to measure, without sacrificing information.

3  
4 Keywords: discriminant analysis, indicators, microbes, military land, soil, vegetation

## 6 INTRODUCTION

7 Resource managers need to be able to document when environments changes occur.  
8 As the complexity of environmental problems becomes recognized, it also becomes clear  
9 that information from only one sector of the ecological system may not be sufficient to  
10 address environmental concerns. The concept of ecological sustainability arose, in part, to  
11 capture the full complexity of features necessary to maintain the services of the Earth.  
12 For example, soil quality indices (Andrews and Carroll 2001) have been devised to  
13 combine various soil data in order to monitor the sustainability of agroecosystems. Many  
14 indicators of sustainability involve information about ecological, social, and economic  
15 aspects of life (e.g., Zhen and Routray 2003). Yet devising ways to monitor changes  
16 within the ecological system also requires a diverse approach.

17 One of the questions that a land managers faces is determining when a particular  
18 land use is so changed in its ecological conditions that the area is becomes another kind  
19 of land use. There is the semantic aspect to this change. For example, foresters have long  
20 argued about what constitutes a forest (is it denoted by basal area, tree density, canopy  
21 cover, or something else). Clearly, when all of the trees are gone, the area is no longer  
22 under a forest. Yet, the land use can still be in forestry if the area is treated as plantations  
23 or left to succeed to trees. Characterizing a particular kind of use has proven to require



1 specific information on that particular condition (e.g., Lugo 1997, Hunter and White  
2 1997). Definitions of soil quality have also proved problematic (Karlen et al. 1997). Less  
3 apparent are what environmental features signify changes in land cover. Resource  
4 managers often seeks such guideposts to indicate the direction, timing and changes on the  
5 land. Much as the brilliant color of autumn leaves in the deciduous forest signal  
6 impending loss of leaves, resource managers would like to know what measurable aspect  
7 of the environment indicate changes in land cover.

8 Indicators are useful for resource managers to document and understand ecological  
9 change. Ecological indicators quantify the magnitude of stress, degree of exposure to  
10 stresses, or ecological response to the exposure (Hunsaker and Carpenter 1990, Suter  
11 1993) and are meant to provide an efficient means to characterize ecological  
12 composition, structure, and function of complex ecological systems (Karr 1981)  
13 Ecological indicators can be used to assess the condition of the environment, to monitor  
14 trends, to provide an early warning signal of changes in the environment, or to diagnose  
15 the cause of an environmental problem (Cairns et al. 1993). Their use assumes that these  
16 indicators reflect environmental changes occurring at various levels in the ecological  
17 hierarchy, from genes to species and ultimately to entire regions (Noon et al. 1999).  
18 Furthermore, tradeoffs between desirable features, costs, and feasibility often determine  
19 the choice of indicators.

20 A challenge in selecting ecological indicators is determining which measures of  
21 ecological systems appropriately characterize the entire system yet are simple enough to  
22 be effectively and efficiently monitored. Ecological indicators often deal with only one  
23 aspect of an environmental system. Here we examine the hypothesis that indicators about



1 the physical features of soil, soil biota, and plants are all necessary to explain changes in  
2 ecological conditions associated with land cover over time. Not only do we consider what  
3 kind of information to include in such an analysis, but also determine how to integrate  
4 data from several sectors (vegetation, soils, and soil microorganisms). The hypothesis is  
5 examined at the Fort Benning military installation in west, central Georgia in the  
6 southeastern United States, an area that historically supported longleaf pine (*Pinus*  
7 *palustris*) and which is now used for military training..

8 While necessary for the preparation of military personnel for combat operations,  
9 training can have a profound impact on the ecological character of military installations.  
10 Military resource managers therefore have the complex task of balancing current and  
11 future training needs with the need to maintain ecological integrity, particularly in cases  
12 where endangered species are potentially affected by training activities. Ecological  
13 indicators provide a means for resource managers to understand changes on the  
14 installation.

15 The objective of this study was to identify indicators that signal ecological change  
16 in intensely and lightly used ecological systems (all Fort Benning has had some  
17 anthropogenic changes). Because the intent was that these indicators become a part of  
18 the ongoing monitoring system at the installation, the indicators should be feasible for the  
19 installation staff to measure and interpret. While the focus is on Fort Benning, the goal is  
20 to develop an approach to identify indicators that would be useful to a diversity of  
21 military installations and other land ownerships (in some cases the actual indicators may  
22 be adopted). Because some environmental impacts may be long-term or may occur after a  
23 lag time, early indications of both current and future change need to be identified. The



1 intent of this identification of indicators is to improve managers' ability to manage  
2 activities that are likely to be damaging and to prevent long-term, negative effects.  
3 Therefore, we hypothesized that a suite of variables is needed to measure changes in  
4 ecological conditions. Hald et al. (2003) found that a combination of indicators was  
5 preferable when used to select grassland sites for successful botanical restoration. The  
6 suite that we considered for military land use included measures of soil properties, soil  
7 microorganisms as a measure of below-ground integrity of the ecosystem, and terrestrial  
8 biological integrity. We have previously described our approach of using a suite of  
9 indicators (Dale et al. 2004) and set forth criteria for selecting indicators (Dale and  
10 Beyler 2001).

#### *Indicators of Soil Quality*

13 Soil quality ["the fitness of a specific kind of soil to function within its  
14 surroundings, support plant and animal productivity, maintain or enhance water and air  
15 quality, and support human health and habitation"(USDA NRES 1996)] has the potential  
16 to be an effective index of ecological change. Sustaining soil quality on military  
17 installations has many important benefits. Enhanced soil quality can help to reduce the  
18 on-site and off-site costs of soil erosion, improve nutrient use efficiencies, and ensure that  
19 the resource is sustained for future military use. It is also essential to maintain other  
20 resources that depend on the soil, such as water quality, air quality, vegetative  
21 productivity, and wildlife habitat. A technology that allows for the easy and cost-  
22 effective measurement and monitoring of soil quality could provide a valuable part of an  
23 overall adaptive land management tool with a goal of sustaining ecosystem health.



Garten et al. (2003) previously demonstrated that measures of soil carbon and nitrogen are ecological indicators that can be used by military land managers to identify changes in soils from training activity and to rank training areas based on soil quality. Greater surface soil bulk density, lower soil carbon concentrations, and less carbon and nitrogen in particulate organic matter (POM) were found at sites with moderate or heavy military use and remediated sites as compared to reference sites at Fort Benning. <<ETC from Chucks abstract – mention those aspects that are shown later in this paper>> In this study, we examined the role of soil quality in relation to soil microbes and vegetation as indicators of land-use change.

### Microbial Indicators

Soil biota are also useful measures of the functional integrity of terrestrial ecosystems. That is, soil microorganisms serve as indicators of the nutrient status and ongoing ecological processes that can potentially predict the future status of an ecological system. Peacock et al. (2001) field tested a methodology that measures shifts in microbial biomass, community composition, and physiological status (measured as microbial signature lipid) as a quick and easy indicator of changes in soil quality as a result of military activities and land management.

No single property is a sole indicator of soil quality, but the collection and analysis of many accepted soil quality indicators can be time-consuming and costly. Biological indicators that measure changes in soil microbial ecology, such as viable biomass, community composition, and physiological status, have been shown to correlate well to changes in other soil quality indicators (i.e., soil porosity, texture, organic matter,



1 phosphorus concentrations, nutrient cycling, and concentrations of elements). Because  
2 changes in these microbial indicators reflect changes in other geophysical properties of  
3 the environment, they may serve as a good overall indicator of soil quality. Therefore, a  
4 methodology that uses these biological indicators would be a useful tool for land  
5 managers to cost-effectively determine changes in soil quality that may result from  
6 human use of the site.

7 We therefore investigated the potential use of signature biomarker analysis for early  
8 detection of changes in soil quality as well as soil quality itself and vegetation indicators  
9 that change as a result of military land use and land management activities. We analyzed  
10 the microbial community response to ecological change by employing lipid biomarker  
11 analyses, specifically phospholipid fatty acid analysis (PLFA). PLFA analysis can  
12 determine community structure shifts within the whole viable microbial biomass in  
13 response to ecological change. The approach of PLFA analysis has been used in habitats  
14 ranging from contaminated groundwater through subsurface and surface soils (Stephen et  
15 al., 1999; MacNaughton et al., 1999) and has provided a comprehensive and detailed  
16 picture of changes in microbial community structure.

### 17 18 *Vegetation Indicators*

19 The major groups of land-cover classification at Fort Benning are partially  
20 determined by past and current land use (Kane and Keeton 1998). For example, there are  
21 few trees in areas of heavy tank traffic. The forest types at Fort Benning have been  
22 defined and mapped by the staff at Fort Benning. Yet the effects of military training on  
23 understory vegetation are less clear. We previously demonstrated that understory plants



1 readily respond to different types of military use (Dale et al. 2002). Our work  
2 demonstrated that the composition, distribution, and density of understory plants were  
3 useful indicators of military use. Of the 137 species that we found in heavy, moderate,  
4 and light use areas, some only appeared in one or two uses. However, we found that  
5 grouping the species by life forms was a reasonable approach for land managers who will  
6 not necessarily experienced in plant taxonomy. In this study, we examined the value of  
7 those vegetation indicators in relation to indicators of soil quality and soil microbiology.

## 9 METHODS

### 10 *Study Sites*

11 Fort Benning is a military base located on 73,503 ha in west central Georgia and a  
12 sliver of eastern Alabama. It serves as the Army's primary training ground for infantry  
13 and is the permanent home of several units, including the 3<sup>rd</sup> Brigade, 3<sup>rd</sup> Infantry  
14 Division (mechanized). Ecologically, it sits on the Fall Line between the Southern  
15 Appalachian Piedmont and Upper Coastal Plain. The climate is characterized by long hot  
16 summers, high humidity, and mild winters. Precipitation is regular throughout the year,  
17 with most occurring in the spring and summer (Knowles and Davo, 1997). Soils are  
18 composed of clay beds, weathered Coastal Plain material, and alluvial deposits from the  
19 Piedmont (Knowles and Davo, 1997).

20 Prior to European settlement, pine forests covered much of the landscape, but since  
21 then much of the pine forests have been lost or degraded (Frost, 1993; Quicke et al.,  
22 1994) mainly as a result of land-use change, timber harvest, and fire suppression  
23 (Haywood et al., 1998; Gilliam and Platt, 1999). Frequent, low intensity fires were once a



1 frequent component of the pine forest ecosystem (Glitzenstein et al., 1995). First Native  
2 Americans and then European settlers farmed some of the area (Kane and Keeton, 1998).  
3 Native Americans occupied the region for thousands of years before European settlement  
4 and had significant impacts on portions of the landscape where they cleared the forest,  
5 burned, and farmed, particularly near to streams. European immigrants settled the area  
6 beginning in the early 1800s, and farming was their predominant land use.

7 The U.S. government began acquiring land in 1918 for the infantry school of arms,  
8 and the permanent military post was established by Congress in 1920 (Kane and Keeton,  
9 1998). Fort Benning is currently used for U.S. military infantry and tank training  
10 exercises, yet it retains large areas in semi-natural vegetation. Fort Benning occurs in the  
11 Southeastern Mixed Forest Province of the Subtropical Division (Bailey, 1995). Today  
12 the installation is characterized by second growth pine forests of longleaf (*Pinus*  
13 *palustris*), loblolly (*P. taeda*), and slash pines (*P. elliotii*) as well as by localized areas  
14 where intensive military training with tracked vehicles has removed most of the  
15 vegetation. Within areas dominated by longleaf pine, resource managers set fires about  
16 every three years reduce hardwood ingrowth that compromises the quality of the habitat  
17 for the federally endangered red-cockaded woodpecker (*Picoides borealis*).

18 Our goal was to develop valid and repeatable indicators of impacts of training on  
19 understory vegetation, soils, and soil microbes in sites suitable for longleaf pine forests.  
20 Thus the study focused on maneuver training areas and does not include firing ranges,  
21 ordinance impact areas, or cantonment areas.

22 Study site locations were on the 65,900 ha of Fort Benning (about 90% of the  
23 installation) with conditions that can support longleaf pine as determined from existing



1 forest stand information (Personal Communication, Bob Larimore 1999, Fort Benning,  
2 GA) and county soil surveys of the United States Department of Agriculture Natural  
3 Resource Conservation Service (USDA NRCS 1924, 1983, 1993, 1997) as described in  
4 Dale et al. (2002). Currently longleaf pine stands occur on approximately 5800 ha of Fort  
5 Benning (USAIC, 2001).

### 6 7 *Sampling Design*

8 A stratified sampling methodology was used to select sites experiencing different  
9 types of military use. Sample sites were blocked into five training intensity categories:  
10 reference, heavy, moderate, and light, and remediation (Figure 1). Reference areas have  
11 little to no training activities and are often in exclusion zones around firing ranges. Heavy  
12 training areas have both wheeled and tracked vehicle training exercises. Moderate  
13 training areas are near tank training zones and are exposed to minor tracked vehicle  
14 maneuvers, as well as limited wheeled vehicle and infantry traffic. Light training areas  
15 experience limited to dismounted training and individual orienteering activities. The  
16 classification of each training type was primarily based on historical records and  
17 generalized maps of training activity. Due to the variability of training intensity over  
18 space, actual site selection involved field reconnaissance with Fort Benning natural  
19 resource personnel.

20 The remediation area is on uplands at McKenna Drop Zone, which was cleared in  
21 1988 and subsequently rehabilitated. The area does not have any current military training  
22 or testing. Restoration of the site included liming, fertilizing, and seeding with a mixture  
23 of grasses and legumes designed to increase vegetative cover and reduce runoff [e.g.,



1 giant reed (*Arundo donax*), Bermuda grass (*Cynodon dactylon*), little bluestem  
2 (*Adropogon scoparius*), Maidencain (*Panicum hemitomom*), Pensacola Bahiagrass  
3 (*Paspalum notatum*), Alamo switchgrass (*Panicum virgatum*), weeping lovegrass  
4 (*Eragrostis curvula*), Lespedeza Sericea (*Lespedeza cuneta*, Var. *Sericia*) and Lespedeza  
5 Interstate (*Lespedeza cuneta*, Var. *Interstate*)].

6 Three sampling transects were randomly located in each of the reference, light,  
7 moderate, and heavy training classifications, and two transects were randomly placed in  
8 the remediation area. Along each of the 14 transects, five circular plots were established  
9 at intervals of 15 m between the centers. The 5 m radius circular plot size was determined  
10 based upon a species-area curve constructed at the reference site. Data on vegetation, soil  
11 microbial composition, and soil quality were collected in September and October of  
12 1999.

13 Within each plot, all species of understory vegetation were identified and visually  
14 assigned a cover class using a modified Braun-Blanquet (1932) cover system (based on  
15 Clarke 1986 p. 64) (Dale et al. 2002), for Bråkenhielm and Qinghong (1995)  
16 demonstrated that visual estimates provide the most accurate, sensitive, and precise  
17 measure of vegetation cover compared to point frequency and subplot frequency  
18 methods. Species identifications were based on Godfrey (1988) and Radford et al. (1968).  
19 Understory species were grouped into the 36 plant families and cover was recorded by  
20 family. In addition, each species was classified using Raunkiaer's five major life forms  
21 (Kershaw and Looney 1985) based on the height of perennating buds. Canopy cover,  
22 understory cover and species richness, and depth of soil A-horizon were also recorded for  
23 each plot. To establish maximum stand age for each transect, we counted the rings from



two tree cores from each of the four largest trees in the immediate vicinity of each transect. This information provided a total of 41 vegetation attributes for each plot.

A surface mineral soil sample was collected from each plot using a hand soil probe (2 cm diameter) to a mean ( $\pm$ S.D.) depth of 22 ( $\pm$ 2) cm. For each sample the microbial community composition was analyzed as described by Peacock et al (2001).

Soil quality measures included particulate organic matter, refractory carbon, and total carbon and nitrogen concentrations and were determined as described by Garten et al. 2003). <<CHUCK DO WE NEED MORE HERE??>>

#### *Stepwise Discriminant Function Analysis*

In order to differentiate among sites and determine which indicators were giving rise to site discrimination patterns at Fort Benning, we used a stepwise discriminant function analysis (Stepwise DFA) (Tabachnick and Fidell 2001). The analysis considered a total of 65 potential indicators: 7 for soil quality, 15 for soil microbes, and 43 for vegetation. For each DFA, 5,000 bootstrap iterations were performed in order to build a sampling distribution against which the observed data was compared.

There are several selection criteria available when performing a stepwise DFA. The criteria are used to determine which variables, from all possible variables, are to be used in the DFA. We chose RAO's V, which is a generalized distance measure that incorporates variables that elicit the greatest change in V or the distance between groups (Tabachnick and Fidell 2001). Thus, variables were selected based on their contribution to the overall separation of groups.



1 The indicator data from Fort Benning were analyzed using the computer language  
2 program Matlab 6.0 ([www.mathworks.com](http://www.mathworks.com)). All Matlab programs used for the analyses  
3 presented here are available at the Texas Tech Biological Sciences website  
4 ([www.biol.ttu.edu/Faculty/FacPages/Strauss/Matlab/matlab.htm](http://www.biol.ttu.edu/Faculty/FacPages/Strauss/Matlab/matlab.htm)). The following Matlab  
5 functions: (a) DISCRIM, (b) STEPDISC, and (c) CLASSIFY were used to conduct  
6 analyses. When employing any stepwise method, it is often desirable to determine the  
7 efficiency of the analysis via a cross-validation procedure. The cross-validation analysis  
8 determines the percentage of observations correctly classified. In essence, a cross-  
9 validation procedure determines the level of resolution the chosen stepping procedure has  
10 in classifying observation to groups. Using the Matlab function CLASSIFY as the cross-  
11 validation procedure, each observation was removed from the data set, and a DFA was  
12 conducted. Afterwards, the removed observation was reclassified to a group based on the  
13 discriminant structure. This approach provides an unbiased estimate for determining to  
14 which group or treatment a particular observation belongs. When the classification  
15 procedure was bootstrapped using the CLASSIFY function in Matlab, an index of percent  
16 correct classification was produced for each site.

17 Once the discriminant functions were calculated, a plot of the stretched attribute  
18 vectors was calculated by rotating the discriminant functions to redistribute the variance.  
19 Correlations between the rotated canonical discriminant functions and each  
20 discriminating variable were calculated and multiplied by the univariate F ratio to  
21 produce (x,y) coordinates for the endpoint of a vector that has its other endpoint at the  
22 origin (Hair et al, 1995).

23



## RESULTS

The model generated four discriminant functions, the first two of which were considered in this analysis. Ten indicators explained 90.1% of the variation between the sites as grouped by military use (Figure 2). The generated model was highly accurate, correctly classifying all but one of the seventy plots (98.6%). As expected, the percentage correctly classified in the cross-validation is marginally lower (95.7%). All classification errors in both model generation and cross-validation occurred for moderate use plots, which might be expected to share some characteristics with plots of both heavy and light military use. Interestingly, the remediated sites fall in the central region of the plot, between the moderate and heavy training sites and the lightly used and reference sites.

The relationships between the individual ecological indicators and the type of military use are apparent from the stretched attribute vectors (Figure 3). Eight of the ten variable vectors have a positive value for DF 1, indicating that their means are greater in the reference and light-use groups, whose centroids are also positive for DF 1. For example, soil carbon content, vegetation cover, and stand age generally tend to be higher in areas with lower military use (Table 1). The bacterial fatty acid indicative of Gram-positive microbes and cover of annual species, however, tend to be higher for areas with moderate and heavy military use as well as the remediated sites. The similarity in six of the vectors suggested that some substitution in indicator choice might be possible.

## DISCUSSION



## Statistical Analyses

The degree to which variation in indicators among sites at Fort Benning were expressed as site specific differences was readily visualized by plots generated from discriminant function analysis (DFA) (Tabachnick and Fidell 2001) (Figures 2 and 3). Since the indicator suite was effective in discriminating differences among sites, samples within sites form distinct clusters in multidimensional space. DFA, when conducted in a stepwise fashion, both decreased the number of variables and incorporated the most pertinent variables for discriminating among sites.

DFA has several unique features: (a) condensation of variables where only a small subset of the total are used in the discriminant analysis, (b) the selected indicators that provide the best separation among sites in question, and (c) clarity of interpretation. Thus, DFA gave rise to patterns that were easily visualized and delimited in discriminant function graphs. In addition, the stepwise discriminant method prevented the occurrence of singularities and multicollinearities that are present when all the indicators are used in the analysis (Tabachnick and Fidell 2001, Stevens 2002). For example, if the number of variables is greater than the number of observations (soil samples), a singular solution is provided by the analysis, which when graphed displays perfect separation among groups. However, that solution is only an artifact of the analysis process, and unfortunately, the scientific literature contains many such faulty analyses. Furthermore, when groups or sites have been identified *a priori* (as in this study), DFA is superior to another commonly used method to describe multivariate data, namely Principal Components Analysis or PCA (Tabachnick and Fidell 2001). DFA has proven useful in other ecological studies. For example, Spooner et al. (2004) used discriminant analysis to



document the effect of anthropogenic activities on the environment. In that case, road activities promoted Acacia structural features.

#### *Ecological implications*

Two outcomes of the analysis are of particular importance. First, the ten indicators selected by the model represent the full range of ecological indicator types that were surveyed in this experiment: vegetation, soil chemistry, and soil microbiology (Table 1). An effective suite of ecological indicators for military resource managers should, therefore, reflect this diversity.

The analysis also demonstrates that many indicators are highly correlated. The vectors for carbon storage, carbon stock, tree age, understory cover, number of understory species, and the cover of plants in the tomato family, as well as the negatively correlated vector for bacterial fatty acid, were separated by only about 20 degrees. Clearly, correlations occur in the ecological functioning of these indicators. For example, McCulley et al. (2004) found that as grasslands and savannas experienced increases in woody plant abundance, that there was an increase in soil C and N accumulation, annual soil respiration, and soil microbial biomass. This information can be used by resource managers to reduce the number of indicators measured without losing information. For example, a manager might elect to measure only understory cover rather than the number of understory species, since it requires less time and taxonomic expertise to visually estimate cover than to tally species. However in selecting indicators, resource managers should take into account not only the ease and expense of measurement but also the features about the ecological system revealed by the indicator (Dale and Beyeler 2001).



1 For example, gram-positive microbes are negatively correlated to most of the vegetation  
2 characteristics, and microbial analysis is expensive, but microbes tend to respond much  
3 more quickly to change than vegetation and soil quality (e.g., Schipper et al. 1994).

4  
5 <<Previous work by Brejda et al. (2000) used discriminant analysis to show the  
6 importance of total organic carbon in differentiated between land uses. Our study is the  
7 first study to look across a suite of indicator types.>>

## 9 CONCLUSIONS

10 Statistical results from this research will allow decision makers to choose among ten  
11 identified indicators to select ones most appropriate for long-term measures of land  
12 changes in longleaf pine forests on military training lands in west central Georgia. We  
13 expect these same indicators may be appropriate for the <ten?> other military  
14 installations that occur along the Fall Line of the southeastern United States. These  
15 particular indicators may also prove of use for other types of land management in the  
16 pine forests of the southeastern United States. The approach of using DFA to differentiate  
17 among a group of indicators for land use condition may be beneficial under other land  
18 uses and for other types of indicators. We hypothesize that such a suite of ecological  
19 indicators may be appropriate in other land uses and locations because of the complexity  
20 typical of ecological systems.

## 23 ACKNOWLEDGMENTS



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## 7 **List of Figures**

8 Figure 1. Representative photographs of each type of land use: (a) reference, (b)  
9 remediated, and (c) heavy, (d) moderate, and (e) light military training.

10  
11  
12 Figure 2. Multivariate analysis of the proposed indicators treating land-use categories as  
13 independent variables.

14  
15 Figure 3. The vector graph with the group centroids plotted. Values that explain 90% of  
16 the variance represent a three sectors of the ecological systems: microbial biology (1.  
17 bacterial fatty acid indicative of gram positive microbes), soil quality (2. carbon storage,  
18 3. carbon stock), and vegetation (4. tree age, 5. canopy cover, 6. understory cover, 7.  
19 number of understory species, 8. cover of annual species, 9. cover of legumes, and 10.  
20 cover of plants in tomato family).



1 Table 1. Mean (and standard error) of ten most significant attributes by land-use  
 2 category. <<AARON – can you provide these values and their units??>>

| Category of attribute | Attribute   | Reference | Light training | Moderate training |
|-----------------------|---|-----------|----------------|-------------------|
| Microbial biology     | Bacterial fatty acid indicative of<br>gram positive microbes  |           |                |                   |
| Soil quality          | Carbon storage<br>Carbon stock  |           |                |                   |
| Vegetation            | Tree age<br>Canopy cover<br>Understory cover<br>Number of understory species<br>Cover of annual species<br>Cover of legumes<br>Cover of plants in the tomato family |           |                |                   |

3

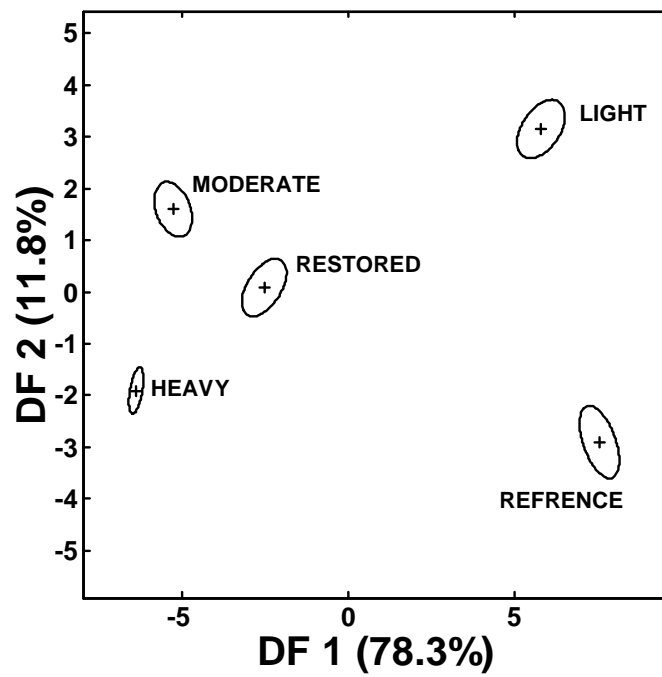


1 Fig. 2 <<NEED TO SPELL "REFERENCE" CORRECTLY>>

2

3

DFA- Centroid (+) surrounded by 95% Confidence ellipse.

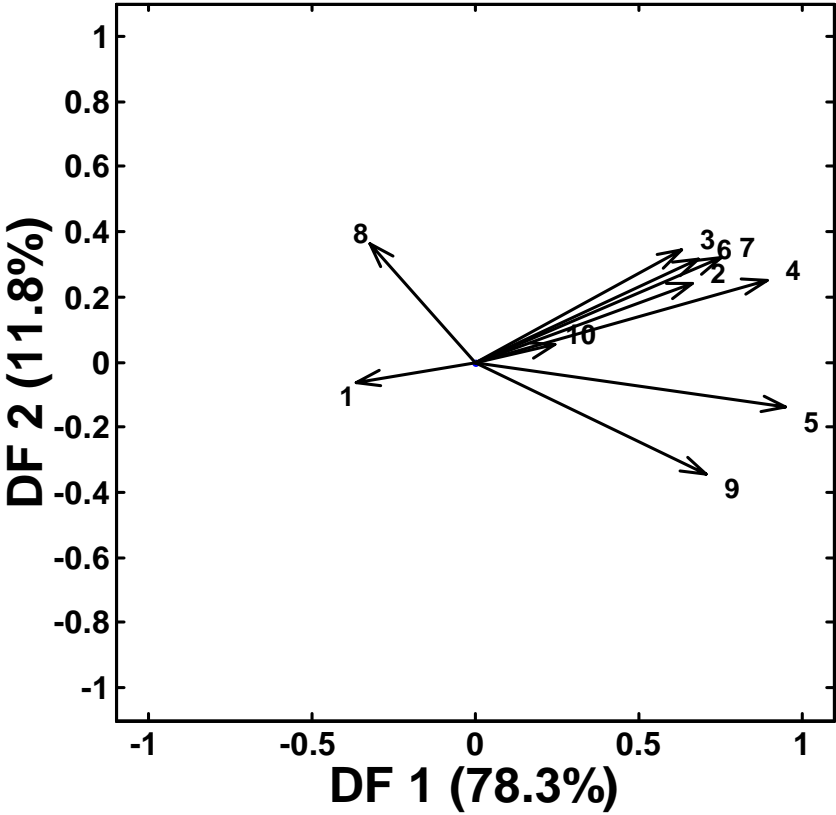


4

5



1    Figure 3



2



## Vehicle impacts on the environment at different spatial scales: Observations in west central Georgia, USA

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### Abstract

Roads and vehicles change the environmental conditions in which they occur. One way to categorize these effects is by the spatial scale of the cause and the impacts. Roads may be viewed from the perspective of road segments, the road network, or roads within land ownership or political boundaries such as counties. This paper examines the hypothesis that the observable impacts of roads on the environment depend on spatial resolution. To examine this hypothesis, the environmental impacts of vehicles and roads were considered at four scales in west central Georgia in and around Fort Benning: a second-order catchment, a third-order watershed, the entire military installation, and the five-county region including Fort Benning. Impacts from an experimental path made by a tracked vehicle were examined in the catchment. Land-cover changes discerned through remote sensing data over the past three decades were considered at the watershed and installation scales. A regional simulation model was used to project changes in land cover for the five-county region. Together these analyses provide a picture of the how environmental impacts of roads and vehicles can occur at different spatial scales. Following tracked vehicle impact with a D7 bulldozer, total vegetation cover responded quickly, but the plant species recovered differently. Soils were compacted in the top 10 cm and are likely to remain so for some time. Examining the watershed from 1974 to 1999 revealed that conversion from forest to nonforest was highest near unpaved roads and trails. At the installation scale, major roads as well as unpaved roads and trails were associated with most of the conversion from forest to nonforest. For the five-county region, most of the conversion from forest to nonforest is projected to be due to urban spread rather than direct road impacts. The study illustrates the value of examining the effects of roads at several scales of resolution and shows that road impacts in west central Georgia are most important at local to subregional scales. The insights from these analyses led to several questions about resource management at different spatial scales.



*Keywords:* Bulk density, Disturbance, Fort Benning, Land cover, Landscape, Management, Scale, Simulation, Soil compaction, Vegetation

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## 1. Introduction

Environmental impacts of human activities vary by spatial scale. In terms of vehicle impacts on the environment, the vehicles themselves and the creation of roads cause the most impact at the fine scale, but broad scale effects can include noise, water or air pollution, and disruption of habitat. Furthermore, ecological systems can be viewed as spatially and temporally hierarchical [1, 2]. In other words, ecological processes observed at one level of organization arise from lower level behaviors and are constrained by higher level processes. As an example, the avoidance of roads by gray wolves (*Canis lupus*) is a fine-scale response that affects broad-scale patterns in wolf density: that is, wolf density is low in areas that have a relatively high density of roads (more than 0.45 km of road per km<sup>2</sup> area) [3]. Thus environmental effects at one scale can, in turn, affect the ecological system at other scales. In this paper we examine the ways in which road effects on the environment can vary by spatial resolution.

Fine-scale environmental impacts that are associated with off-road vehicle movement include soil compaction and changes in vegetation properties such as species, cover, and diversity in association with crushing and later plant colonization and competition. Studies of tracked vehicle impacts on vegetation at military installations in semiarid and arid environments have demonstrated changes in soil compaction [4], herbaceous plant composition [5, 6], density, and cover [7]. Increased soil compaction would lead to longer recovery periods for the affected plant properties. The level of effect on vegetation is determined by the exact path of the vehicle: sharp turns by tracked vehicles disturb a larger width of soil and cause deeper track ruts than smooth turns or straight operation [8].

Local environmental effects such as changes in soil bulk density and vegetation can, in turn, cause regional problems. The introduction and subsequent spread of introduced species can lead to broad-scale land-cover changes. For example, Scotch broom (*Cytisus scoparius*) was planted along selected highways in western Washington state for beautification but has now become a regional pest that competes with native species, widely disseminates pollen to which many people are allergic, disrupts fire regimes, and provides habitat for feral animals [9]. Kudzu (*Pueraria lobata* Ohwi) is another example of a deliberately introduced plant that has become a regional pest along roads. Kudzu was established in the southern United States for erosion control, as fodder for cattle and sheep, and as a porch vine. This liana native of China has overgrown and killed trees in many locations in the American Southeast [10]. In addition to quickening the spread of such invasive species, road development can alter surface water bodies by changing wetland drainage, forcing streams into channels, and increasing inputs of sediment, road salt, and heavy metal to streams [11]. Thus, the cumulative effects of local events can lead to regional changes.

Land use and land management are basically local phenomena: farmers clear land for crops; state governments construct roads between cities; businesses are developed in industrial parks. Nevertheless, across the globe most land transformations now have large-scale effects although they originate from local changes. As an example of



cumulative effects, timber harvesting and clearing occur at local scales but, when aggregated, can result in large-scale deforestation [12]. Moreover, Shaw and Diersing [5] speculate that the impacts of tracked vehicles on the density and cover of woody plants at the local scale at a military installation in Colorado could exceed a threshold for sustainability of larger scale juniper (*Juniper monosperma*) woodlands. If the trend of reductions in density and cover of woody plants continues, the density of juniper, which dominates the woodlands, would be reduced to a critical level, for regrowth of this species is very slow. Local versus broad-scale perspectives on the benefits and costs of land management provide different views of the implications of land actions. Recognition that human impacts occur on a broad scale as well as a local scale is changing the way that natural resources are managed.

A multi-scale perspective is needed to address today's land management problems [13] for several reasons. It is now recognized that the spatial scale of environmental problems is complex and can be multifaceted. Furthermore, all ecological processes (and management actions) occur in a spatial context and are constrained by spatial location. A broad-scale perspective is necessary for the management of wide-ranging animals [e.g., the Florida panther (*Puma concolor coryi*) [14] or the marbled murrelet (*Brachyramphus marmoratus*) [15] ]. Understanding and managing disturbance also requires a broad-scale perspective because land-cover patterns can retard or incite the spread of natural or anthropogenic disturbances (e.g., connected forests may lead to larger fires). Therefore, solutions for contemporary environmental problems need to be provided within a spatial context. For example, natural areas that provide essential ecological services (e.g., cleansing of water) are limited in extent, and their contributions must be interpreted within the landscape matrix in which they occur and with the understanding that environmental conditions may change spatially or across an area as well as over time (as with global warming). Thus, spatially optimal solutions to land management problems should be considered.

Military installations are an ideal setting in which to examine the environmental impacts of vehicles and roads at multiple scales. Military training involves the use of tracked and wheeled vehicles in off-road locations, as well as on unpaved and paved roads, and new roads are periodically constructed to provide access to new training areas. The military installations themselves can be situated near highway development and urbanization. Road densities commonly are high around military bases because civilian traffic is not permitted through much of the installation area, which forces that traffic to occur near the perimeter or in specified corridors.

As with any area where off-road vehicles operate, their operation on military installations impact soil, vegetation, and streams in the immediate area of vehicle operation; whereas erosion and noise may cause impacts at a distance; and land-use change along roads can result in still other cumulative environmental effects. Effects of vehicles or roads on vegetation or wildlife at military installations are often environmentally significant because these installations serve as reservoirs for vegetation diversity and for threatened and endangered species [16].

The question considered in this paper is how the environmental impacts of roads vary by spatial resolution. We hypothesized that impacts of road segments at fine scales would largely be to local soil and vegetation conditions and that impacts of road networks would be observable at the broad scale, i.e., conversion from the native forest cover types



to nonforest conditions. We anticipated that as the scale of resolution became more broad, the effects of roads would be more pronounced at a distance. To examine this concept, environmental impacts of vehicles and roads were considered at four scales in west central Georgia in and around Fort Benning. Field experiments, comparison of land cover over time (as determined through remote sensing analysis), and a simulation model were used to determine potential environmental effects. The finest scale was a single second-order catchment (4 ha) within a training compartment in the northeast corner of Fort Benning, referred to by the installation's land managers as compartment K-11. An experimental disturbance was created in the catchment with a D7 bulldozer. This catchment was also thinned and burned as part of routine management prior to the tracked-vehicle disturbance. The second scale was a third-order 244 hectare watershed also in training compartment K-11 of Fort Benning. The third scale was the entire 73,503-hectare Fort Benning installation. The fourth, or broadest, scale was a five-county region in west central Georgia (Harris, Talbot, Muscogee, Marion and Chattahoochee counties) of 442,347 ha containing Fort Benning, the city of Columbus, and extensive farm and forest land primarily in private ownership. The insights from these analyses led to several questions about resource management at different spatial scales.

## **2. Methods**

### *2.1 Site Description*

The climate of the study area in west central Georgia is characterized by long, hot summers and mild winters, and precipitation is regular throughout the year but with most occurring in the spring and summer [17]. Soils are composed of clay beds, weathered Coastal Plain material, and alluvial deposits from the Piedmont [17]. Before the military base was established in 1918, both Native Americans and European settlers farmed the region largely growing corn and cotton, respectively [18]. Fort Benning is currently used extensively for U.S. military infantry and tank training exercises, yet it retains large areas within the installation in semi-natural vegetation. Fort Benning constitutes part of the Southeastern Mixed Forest Province of the Subtropical Division [19], which is now characterized by second-growth pine forests of longleaf (*Pinus palustris*), loblolly (*P. taeda*), and slash pines (*P. elliotii*) mixed with many species of oaks and other deciduous trees. Frequent, low-intensity fires are thought to have been an integral component of the pine forest ecosystem [20] and have been a component of the management plan at Fort Benning since the 1970s. Before European settlement began, pine forests covered much of the landscape, but since then they have been lost or degraded [21, 22] mainly as a result of land-use change, timber harvest, and fire suppression [23, 24].

### *2.2 Approach for each scale*

#### *2.2.1 Local scale*

Because tracked vehicles can cause environmental damage to plants and soil [8], attributes of both these features were measured after a disturbance. In May 2003, a disturbance treatment was created within an experimental catchment in training compartment K11 at Fort Benning. Several passes of a D7 bulldozer with the blade lowered were used to remove both extant vegetation cover and surface soil organic matter. Vegetation surveys were conducted shortly after the disturbance treatment in June and in September to capture the temporal response in plant cover. Three sets of 50-



m transects were established to monitor response and recovery from the disturbance. Control transects were established parallel to the disturbance treatments at a distance of 5 m. Ten points were chosen at random along each treatment and control transect, for a total of 60 survey points, and plant cover was assessed using 0.568-m radial plots at each point. Total and individual species plant cover was ranked according to a modified form of the Braun-Blanquet [25] cover-abundance system [26] (Table 1). Species identification followed Radford et al. [27]. Matlab ® [28] was utilized for data analysis.

Replicate soil samples were collected at the randomly chosen sampling points along both treatment and control transects to a depth of 30 cm by means of a soil probe (2.54-cm diameter) with hammer attachment (AMS, American Falls, ID) in June 2003. The O-horizon, when present, was removed from a known area (214 cm<sup>2</sup>) prior to sampling the mineral soil. The mineral soil samples at each sampling point were cut into 10-cm increments and composited by depth. Soil density was calculated on the basis of air-dry mass (<2 mm) and the known volume of the sample. O-horizon mass was determined after oven-drying (75°C).

### *2.2.2 Watershed scale*

The third-order watershed in training compartment K11 was selected for analysis at an intermediate scale. This watershed has not experienced major tracked vehicle traffic, but it does have several unpaved roads and trails. Orthophotographs and current roads maps were used to determine how the roads within the watershed had changed since the 1950s.

Changes in land cover over time from the 1970s to the 1990s were assessed through the use of satellite imagery [29]. A combination of ARC INFO 7.2.1™, GRID™, ArcView 3.2™, and ERDAS IMAGINE 8.2™ software was used to derive land cover from satellite imagery. The North American Landscape Characterization (NALC) data that are largely derived from Landsat Multispectral Scanner (MSS) imagery were used in this analysis. The NALC data have a sample resolution of 60 m. The NALC data set covering the Fort Benning area is composed of triplicates dated 1974, 1983/86, and 1991 for two scenes (i.e., path 019/row 037 and path 019/row 038). The two scenes for each time period had to be connected in a mosaic in IMAGINE™ before the classification process could begin. The two scenes comprising the mosaic for the 1980s were made in different years; however, given the nature of the landscape and method of comparison used, this time interval was considered acceptable, and the date of mosaic is referred to as “1983.” Two Landsat-7 Enhanced Thematic Mapper (ETM) images dated July 24, 1999, were used to create a current land-cover map of Fort Benning.

Unsupervised classification, which identifies a user-defined number of classes based upon spectral response, was used to create 45 spectral classes from the imagery. These 45 classes were then combined into six land-cover classes with the use of a 0.5-m resolution digital color orthophoto from 1999 and Land Condition Trend Analysis (LCTA) [30] point data of 1991 as reference data. The six classes are water, barren or developed land, pine forest, deciduous forest, mixed forest (deciduous and pine, areas of sparse forest cover, or areas of transition between forest and nonforest), and cleared lands (areas cleared of forest vegetation but with some ground cover that may be grass or transitional areas). For comparison with data derived from other imagery sources for



other years, the unsupervised classification of the 1999 image (measured at 30-m resolution) was resampled to a 60-m resolution by means of nearest neighbor resampling.

Post-classification change detection was conducted for the land-cover maps derived from the NALC data. Two operations were carried out to identify the influence of roads on the land cover. First, the changes from forest categories to nonforest categories for the watershed as a whole were identified. The forest categories include deciduous, evergreen, and mixed forests, whereas the nonforest category includes cleared and barren land. Through map queries in ArcView 3.2™, locations of regions belonging to a forest category in an earlier year but to a nonforest category in a later year were identified. By this approach, the percentage of change from forest to nonforest was calculated for the time period 1974 to 1991.

The second process to evaluate road and vehicle influence involved quantifying the forest-to-nonforest conversion at various distances from the roads. Only unpaved roads and tank trails occur in the watershed. Buffers were created on the land-cover maps at distances of 60, 120, 180, 240, and 300 m from the roads within the watershed. Multiples of 60 were chosen because the pixel resolution of the land-cover map was 60 m. The buffers were used to extract regions of forest-to-nonforest change within the specified distances. Based on the number of pixels that changed from forest to nonforest between 1974 and 1991, percentages of change were calculated. This process was carried out for each of the buffer distances.

### *2.2.3 Installation scale*

Land cover for all of Fort Benning was derived as described for the watershed (sect. 2.2.2). Change detection for the entire installation was performed by identifying the percentage change from forest to nonforest for three time periods: 1974 to 1983/86, 1983/86 to 1991, and 1991 to 1999. Land-cover maps generated from the NALC data set and the Landsat ETM images were used for this purpose. Six classes, as described for the watershed scale, were used. The change detection process for the installation scale was similar to that for the watershed-scale (i.e., by means of map queries in Arc View 3.2™). For the entire installation, the road buffers were created for three types of roads: major roads (two- and four-lane highways, including interstates), minor paved roads, and unpaved roads and trails. The forest-to-nonforest conversion buffer analyses were carried out separately for each road type.

### *2.2.4 Regional scale*

The analysis of road impacts for the region was based on a computer simulation model, the Regional Simulator (RSim), which is described in detail elsewhere [31, 32]. As with the watershed and installation scales, land cover was the subject of analysis, but more specifically the effect of roads on urbanization within the region was assessed. The region for the simulation consisted of five counties in west central Georgia: Harris, Talbot, Muscogee, Chattahoochee, and Marion. This area encompasses the middle reach of the Chattahoochee River basin; the Columbus, Georgia, municipality and smaller communities; agricultural, forest, industrial and residential lands; and most of Fort Benning. The output of RSim includes projected maps of land cover for different time steps. The RSim model was developed for the region around Fort Benning, Georgia, because of the large amount of data available for the installation and surrounding region



and because of the cooperation offered by the installation in developing and testing the model. However, the model is being designed so that it is broadly applicable to environmental management concerns for other areas as well.

The urban growth submodel in RSim consists of spontaneous growth of new urban areas, patch growth (growth of preexisting urban patches), and road-influenced urbanization constraints that are applied at each iteration of the model to create new urban land cover [31, 32]. This approach builds upon the concepts set forth by a regional planning model called SLEUTH [33, 34, 35, 36]. Spontaneous urban growth in RSim allows for randomized urbanization, and the patch growth in RSim is influenced by the proximity of existing urban centers. Road-influenced urban growth considers the proximity of major roads to newly urbanized areas. Upon each iteration of the urban growth model, a set number of non-urban pixels in a land-cover map are tested for suitability for urbanization according to the spontaneous and patch growth constraints. For each pixel that is converted to urban land use, an additional test is performed to determine whether a major road is within a predefined distance from the newly urbanized pixel. The proposed road changes were primarily derived from the Georgia Department of Transportation's (DOT's) Governor's Road Improvement Program (GRIP) [37], which began in 1989 and plans to widen two-lane roads to four-lane roads and to attract economic development by improving the state's highway network.

In order to identify a candidate road for growth, a search procedure is performed in RSim to seek out potential pixels for urbanization [31]. The search process continues either until it must be aborted because a suitable direction is lacking or until the distance traveled exceeds a predefined travel limit coefficient. To simulate the higher costs of traveling along smaller two-lane roads than along larger four-lane roads, each single-pixel advancement on a two-lane road contributes more toward the travel limit than a single-pixel advancement on a four-lane road; this accounting in effect allows longer searches along four-lane roads.

Upon the successful completion of a search, the immediate neighbors of the final road pixel visited are tested for potential urbanization. If a candidate pixel for urbanization is found, it is changed to an urban type and its immediate neighbors are also tested to find two more urban candidates. If successful, this process creates a new urban center that may result in spreading growth as determined by the patch growth constraint.

### 3. Results

#### 3.1 Local scale

Total plant cover was substantially lower in the treatment transects following the bulldozer disturbance in June; however, this difference was no longer apparent by September, when both control and treatment transects had the same median cover values (Figure 1A). Yet, not all plant species responded in the same way as did total vegetation cover. The median cover category for juniper leaf (*Polypremum procumbens*) in June for both the control and treatment transects was 0, but in September the treatment transects displayed greater cover than the control transects (Figure 1B).

As expected, O-horizon mass was significantly reduced along the treatment transect ( $257 \text{ g m}^{-2}$ ) in comparison with the control transect ( $640 \text{ g m}^{-2}$ ) ( $F_{1,26} = 9.41$ ,  $P < 0.01$ ). Thus, the treatment caused a substantial reduction in forest floor organic matter. Surface (0-10 cm) soil density under the treatment transect was significantly greater than that under the control transect ( $F_{1,12} = 6.48$ ;  $P < 0.05$ ) (Table 2). The mean ( $\pm$ SE) densities of surface soil samples from the treatment and control transects were  $1.43 \pm 0.03$



and  $1.28 \pm 0.06$ , respectively. Although soil densities for increments deeper than 10 cm tended to be greater under the treatment transect, the differences were not significantly different from the controls. Soil compaction from the bulldozer track at K11 was primarily limited to the surface mineral soil layer and produced an increase of approximately 12% in surface soil density.

### *3.2 Watershed scale*

Road effects within the watershed over the period from 1974 to 1991 were quantified by examining conversion of cover types from forest to nonforest within buffered distances from roads. Visual comparison of the orthophotographs determined that the roads in the study watershed had been there since before the 1960s. This result made it possible to analyze the effect of roads within the watershed in the given time period. The 7.2 km of unpaved roads and trails was used to create buffers and to identify changes in forest cover over the 25-year period from 1974 to 1999 (Figure 2). In general, land-cover conversion tends to decrease as the distances from the roads increase. The land closest to the roads (0 to 60 m) showed a 35% conversion from forest to nonforests, and as the distance increased, the percentage of conversion was reduced. At 120 to 180 m from the roads, 21% of the land was subject to conversion, which is close to 21.9%, the overall percentage of conversion of forest to nonforest in the watershed. However, when the distances increased further, edge effects started to show up and the influence of roads in adjacent watersheds played a role. A visual analysis of the nearby roads on the map clarified this effect. To prevent such effects, the buffer distance for analysis was restricted to 300 m.

### *3.3 Installation scale*

Change detection performed over the installation provided percentages of change from forest to nonforest categories over different time periods (Figure 3). The first time period, from 1974 through 1983/86, showed a slight decline, but there was an overall increase by 1991. Part of this difference may be attributed to the differing lengths of the two time frames. In the first case a longer span, 10 to 13 years, was considered, and in the second case a shorter span, 6 to 9 years, was considered. Most of the change occurred in the third period, from 1991 to 1999. This last period is only 9 years, so it does not fit the observation that the comparative length of the first two periods affected the amount of change.

It is not known how the roads for the entire installation changed over the years of the analysis; however it is assumed that no major changes occurred. Therefore, the data layer of roads present in 1995 was used to create buffers and to identify changes over the years. The buffer analysis carried out for the watershed scale differs from that carried out for the installation scale in that many more types of roads exist at the installation scale. The 316 km of major roads consists of interstate and two- and four-lane highways, which cut across the installation. The largest effect of these major roads was the 18.3% change from forest to nonforest for the 0- to 60-m buffer, with the effect being stable for distances greater than 60 m (Figure 4A). The 148 km of minor paved roads had a smaller (13.6%) effect on forest conversion at the 0- to 60-m distance. Observations of the maps suggest that, at a buffer distance of 120 to 180 m, forest conversion near minor paved roads was influenced by the major roads (Figure 4B). Thus the 1,568 km of unpaved



roads and trails had a large effect on forest conversion, with the percentage of forest conversion declining from 19% to 9% as the distance from the road increased from 0-to-60 m to 240-to-300m (Figure 4C).

### 3.4 Regional scale

Urban growth predictions generated by the RSim model results in an increase of 6.8% of pixels that are in an urban land-cover category under conditions expected to prevail in the coming decades (Figure 5). Most of these new urban areas are near Columbus. Where Columbus is close to the northern boundary of Fort Benning, the projected growth is directly adjacent to military lands.

The regional simulation model projected few urban growth differences between maps produced with the influence of new roads and maps produced without the influence of new roads even though a road-instigated urbanization algorithm was a part of the model. Most of the change to urban land cover resulted from the spread of the urban areas, and less than 0.2% of the total change in 30 simulation steps for the five county region could be attributed to the influence of roads.

## 4. Discussion

### 4.1 Local catchment

The response of total plant cover to mechanized disturbance shows a remarkable recovery by 4 months after the disturbance to an equivalent value of the control transects. The rapid recovery occurred over the growing season and during an abnormally wet summer [38], even though soil compaction was certainly of longer duration. Despite the renewed cover, however, vegetation composition became significantly different from that in the control transects. The September survey showed a significant increase in juniper leaf (*Polypremum procumbens*). This species-specific increase agrees with the species ecology described by Radford et al. (1968), who notes that *P. procumbens* is found within habitats showing recent disturbance, including roadsides.

Two plots (DN2-4 and DN2-6) frequently appear as outliers in Figure 1. These plots were not directly impacted by the blade of the bulldozer but were located between the path of its tracks as it moved between transects. As a result, their composition reflected partial disturbance. The other outliers most likely were a result of environmental heterogeneity created either during the disturbance (DI1-20) or preceding it (DI3-3).

Rapid recovery from soil compaction is not expected. Studies of military training on dry sandy soils indicate that surface soil compaction caused by heavy tracked vehicles can persist for decades [39]. Soil compaction can change the properties of soil pores affecting infiltration capacity [39], the accessibility of organic matter to soil microorganisms, organic matter decomposition rates, and soil N availability [40]. Soil compaction by heavy machinery is also detrimental to root development and plant growth [41, 42, 43]. Soil compaction is a potential long-term effect of heavy vehicle use, and it can have an overall adverse impact on soil and vegetation properties.

Previous studies along disturbance gradients at Fort Benning [44] indicate a persistence of soil compaction for several years following site disturbance. The persistence of soil compaction depends on both soil clay content and moisture status at the time of disturbance. Fine-textured or wet soils are more prone to compaction by heavy vehicle traffic than coarse-textured or dry soils [41, 45, 46, 47], but shrink/swell



cycles in soils with significant clay content [46] or repeated cycles of soil wetting and drying [48] or ecological succession [49] can act singularly or together to reduce soil compaction over long time periods.

#### *4.2 Watershed scale*

Forest conversion was highest near unpaved roads and tank trails in the K11 watershed. Since the roads have width of about 3.7 m per lane, the 0- to 60-m buffer includes the roads themselves. The remote-sensing evidence suggests that at the watershed scale, vehicles on the unpaved roads and the roads themselves affected the areas closest to them. In this context, closeness can be defined as a buffer distance of approximately 120 m. Within that zone, clearing of trees and road-bed erosion likely caused many observed changes over the 25-year period.

#### *4.3 Installation scale*

Major roads and unpaved roads and trails were associated with most of the conversion from forest to nonforest cover at the scale of the installation. Within Fort Benning, these types of roads cover a larger area than the minor paved roads. In addition, the forest conversion in the buffered area near the major roads and unpaved roads and trails declined as distance increased. Along some major roads, considerable clearing (especially along the western edge of Fort Benning) had taken place. Because this western area makes up the cantonment where soldiers live and work, most of this conversion was likely associated with urban growth and expansion.

The minor paved roads graph is bimodal, with a peak at the 0- to 60-m buffer and a peak at the 120- to 180-m buffer (Figure 4B). This second peak could result from the paucity of paved roads and their proximity to major roads. The large forest conversions near major roads (Figure 4A) could have affected the minor paved roads, for the two are often close to each other at Fort Benning.

#### *4.4. Regional scale*

Most of the urban growth is projected to result from urban spread rather than road impacts (Figure 5). Columbus is a rapidly developing municipality, and its high urban growth trend is simulated in the RSim model. Furthermore, by 2004 the Governor's Road Improvement Program (GRIP) planned for the next 25 years will have completed its major activity in the five-country region of this study. Yet roads uniquely produce linear features, which can act to dissect a connected landscape. Effects of roads on ecological pattern and connectivity, as well as effects on nonurban land-cover types, have yet to be examined in the five-county region, and RSim can facilitate such analyses.

#### *4.5 Scale of vehicle and road impacts on the environment*

Table 3 illustrates our hypothesis that road impacts differ by scale and have unique effects on the environment at each resolution. At the resolution of a road segment, there are pressures for establishment and use of roads, which can result in vegetation removal and soil compaction. At the larger watershed scale, the need for military training within the installation calls for roads that can be used for maneuvers with the result of local conversion of forest to nonforest land. Similar pressures and effects occur at the installation scale, but the restrictions of the Endangered Species Act influence



management decisions. As a federal facility, habitats for federally listed species must be protected, which limits the extent and places where military training can occur. At the resolution of the five-county area, the pressure for urban development appears to have a more pronounced impact on conversion of forest to nonforest land than the roads themselves. Of course, road development and improvement are a part of urban expansion, but it appears that change in land use is the prevailing influence on forest conversion for the region. As a largely local phenomenon, road establishment and use may have greater impacts at local and subregional scales, and effects at regional scales may be overridden by other pressures and processes. The concept is supported by the “road effect zone” that is based on observational evidence that environmental effects can extend as far as 1 km from a road [50]. At a national perspective, this road-zone effect translates into about one fifth the area of the United States being affected by roads [51]. Even so, there are large areas where roads are not the primary influence on environmental conditions as well as locales where road effects are pervasive. Our analysis from west central Georgia suggests that road effects should be considered at local and subregional resolutions.

In summary, the results of these combined studies for Fort Benning suggest that effects at all scales are important to consider. Even at the broadest scale of the five-county region, it is the relative relationship between urban growth and the influence of roads that helps to determine the importance of road impacts. The mid-scale remote-sensing analysis suggested that forest conversion was greatest nearest the roads. A local tracked vehicle impact study demonstrated that vegetation cover might not be indicative of the full recovery of mature vegetation or of soil compaction. Hence field studies, remote-sensing analyses, and modeling all have their place in understanding environmental impacts of vehicle and roads.

## **5. Management questions**

Roads on military lands are unique because of the high number of unpaved roads and trails that are heavily used by tracked vehicles [52]. Even so, military lands support a high number of rare species and their habitats [53]. It is partly for this reason that roads on military lands have received special attention. Yet, many questions still remain about appropriate ways to manage for ecological impacts of roads on and near military lands. For example, it would be useful to catalogue the road features that are unique to military lands and those that are common to other types of land ownership or use.

A key question resulting from this multifaceted study is: what metrics should be used to assess road impacts on ecological systems? In this study we used different techniques for determining potential impact at each scale of analysis. At the local scale, we examined the total percentage of plant cover, cover by species, and soil density of different depths. At the watershed and installation scales, past forest conversion in relation to distance from roads was used as a metric. Simulated urbanization was examined at the regional scale.

Instead of these techniques and metrics, we could have used other alternatives. For example, groups of species may respond similarly to roads and traffic at the local scale [54]. In addition, historical orthophotography can be used to create a time sequence of data layers of road for the entire installation, and the developing road network can be used to estimate how much forest conversion is influenced by distance from roads at each time period. In addition, the simulation approach can be refined to explore not only the



causes but also the impacts of road-induced urbanization. For example, models can be used to determine how road infrastructure can affect changes in noise, air, and water quality as well as habitat alteration.

Features of roads themselves can be used as metrics of environmental impacts. Such metrics as the number of passes of a tracked vehicle, length of paved road, number of times a stream is crossed per unit of road length, or road width (e.g., two or four lane) all contain information on how roads and vehicles can impact the environment.

Further studies are needed to attribute causes to effects on land cover. Experimental studies to attribute causality could most easily be carried out at the local scale. At that scale, for example, the relationship between vegetation diversity and cover, on one hand, and soil compaction, on the other, can be explored. The mechanistic relationships between soil compaction and growth and yield of woody plants are reviewed by Kozlowski [55]. Unfortunately, causes of forest conversion near roads at larger spatial scales are difficult to identify through retrospective assessment. As stated in the introduction, the clearing of trees and roadbed erosion are both likely contributors to vegetation impacts; however, the direct crushing of vegetation by tracked vehicles and compaction of soils along roads surely occurred regularly in the past. Records of military activities in road corridors and experiments at the local scale would prove useful for determining causes (or at least reasonable hypotheses) for forest conversion; however such records are not available.

Addressing measurement needs and attributing causation to land-cover change would help determine how environmental impacts might be avoided or mitigated. Considering the metrics in terms of spatial resolution will help assess conditions under which spatial scale might make a difference to management options. In any case, a suite of approaches and metrics likely will best reveal how vehicles and roads affect their environment.

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Table 1. Key of the Cover-abundance class modified from the Braun-Blanquet [24] system.

| Cover-abundance class | Species cover and distribution characteristics |
|-----------------------|--|
| 0                     | No plants present                              |
| 1                     | Less than 1% cover; 1-5 small individuals      |
| 2                     | Less than 1% cover; many small individuals     |
| 3                     | Less than 1% cover; few large individuals      |
| 4                     | 1-5% cover                                     |
| 5                     | 5-12% cover                                    |
| 6                     | 12-25% cover                                   |
| 7                     | 25-50% cover                                   |
| 8                     | 50-75% cover                                   |
| 9                     | 75-100% cover                                  |

Table 2. Mean ( $\pm$ SE) soil densities ( $\text{g}/\text{cm}^3$ ) with depth under treatment and control transects. Each mean is based on seven measurements.

| Soil depth (cm) | Treatment        | Control          | Probability* |
|-----------------|------------------|------------------|--------------|
| 0-10            | 1.44 $\pm$ 0.026 | 1.28 $\pm$ 0.058 | <0.05        |
| 10-20           | 1.63 $\pm$ 0.030 | 1.56 $\pm$ 0.042 | NS           |
| 20-30           | 1.68 $\pm$ 0.027 | 1.61 $\pm$ 0.048 | NS           |

<\*NS = not significant>



Table 3. Four spatial resolutions and their pressures on roads with corresponding effects on roads and the environment.

| Road resolution                        | Area (ha) | Pressures                      | Effect on roads   | Effects on the environment                                 |
|--|-----------|--------------------------------|---|--|
| Road segment in second order catchment | 4         | Establishment and use of roads | New roads   | Vegetation removal and soil compaction                     |
| Road network in third order watershed  | 244       | Military training              | Roads for training  | Local conversion of forest to nonforest                    |
| Road network within Fort Benning       | 73,503    | Military training              | Roads for training  | Conversion of forest to nonforest over entire installation |
|  |           | Endangered Species Act         | Must retain habitat for listed species when using federal funds | Protection of habitat for listed species                   |
| Road network in five county area       | 442,357   | Changes in land use            | Land cover change<br><br>Pressure for new and improved roads    | Regional conversion of forest to nonforest                 |



Figure 1. Surveys within control, C, and treatment, T, transects in June and September of (A) total plant cover, and (B) cover of *Polypremum procumbens*. In these box plots, the median is represented by a solid line; the 25<sup>th</sup> and 75<sup>th</sup> percentiles, by the upper and lower edges of the box; and the minimum and maximum values of the data, by the dashed lines. Outliers, values more than 1.5 times the box extent, are shown with a circle.

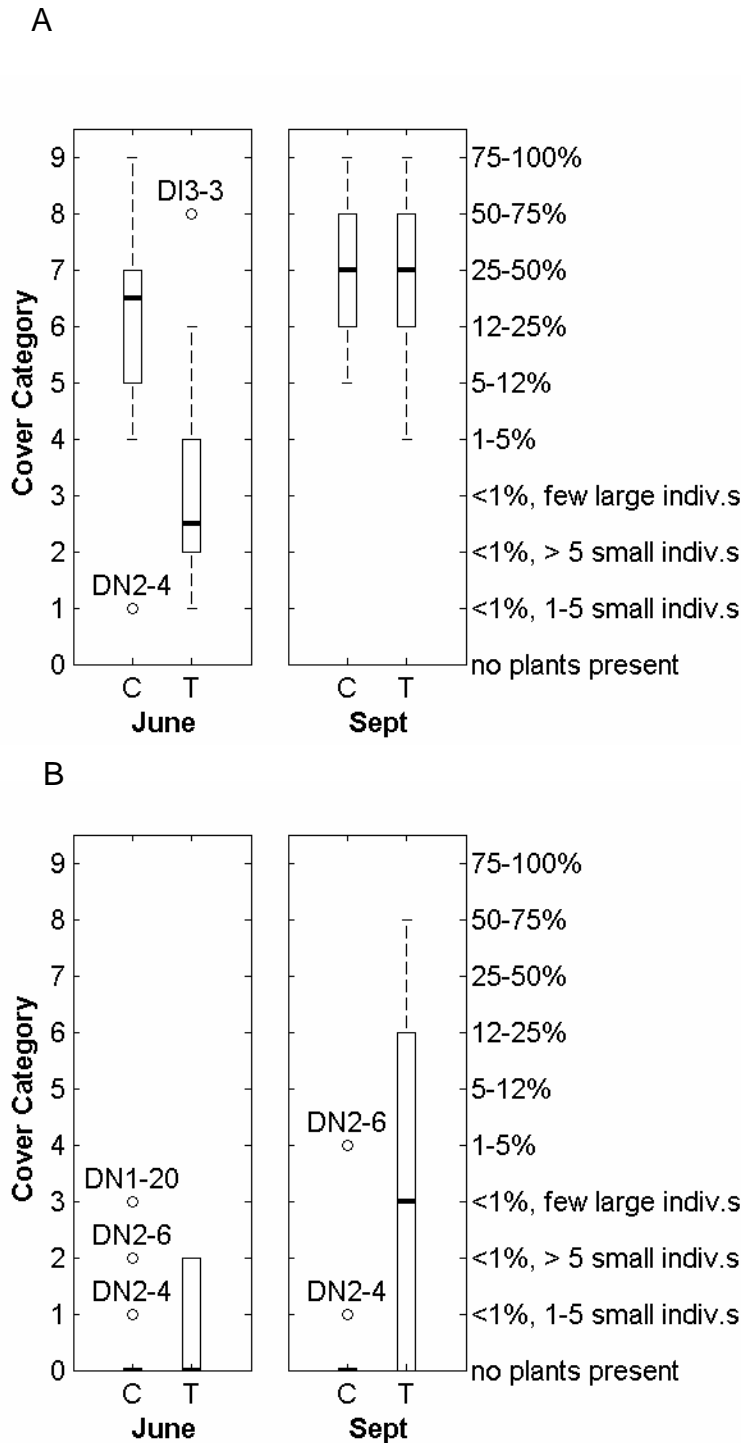




Figure 2. Percent change in the conversion from forest to nonforest from 1974 to 1999 at different distances from unpaved roads and tank trails for the study watershed in training compartment K11.

### Percentage

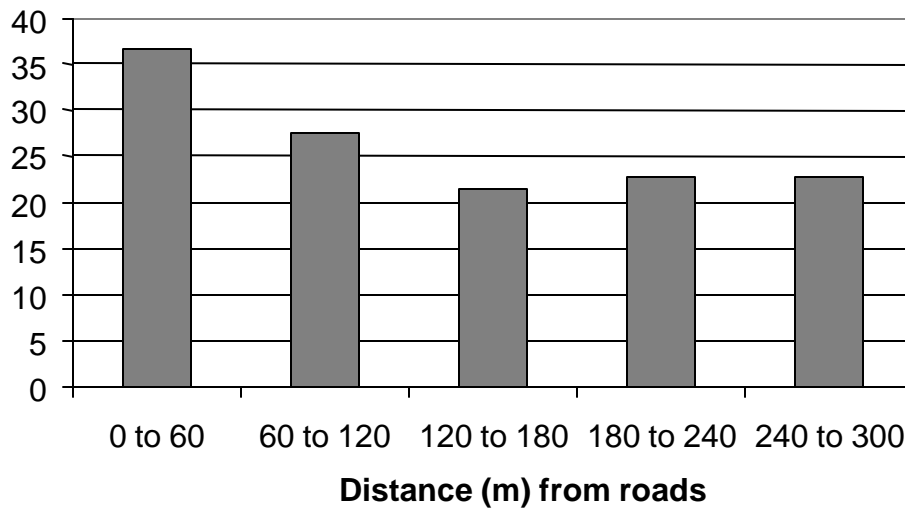


Figure 3. Results of the change detection performed on the land-cover maps of Fort Benning. The percentages indicate conversion from forest to nonforest land-cover categories for different time periods.

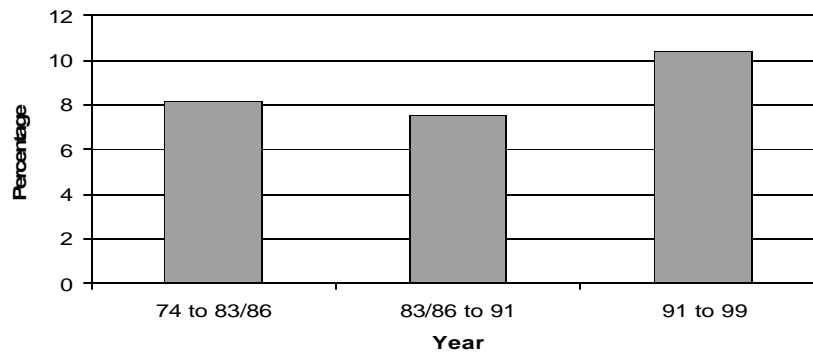
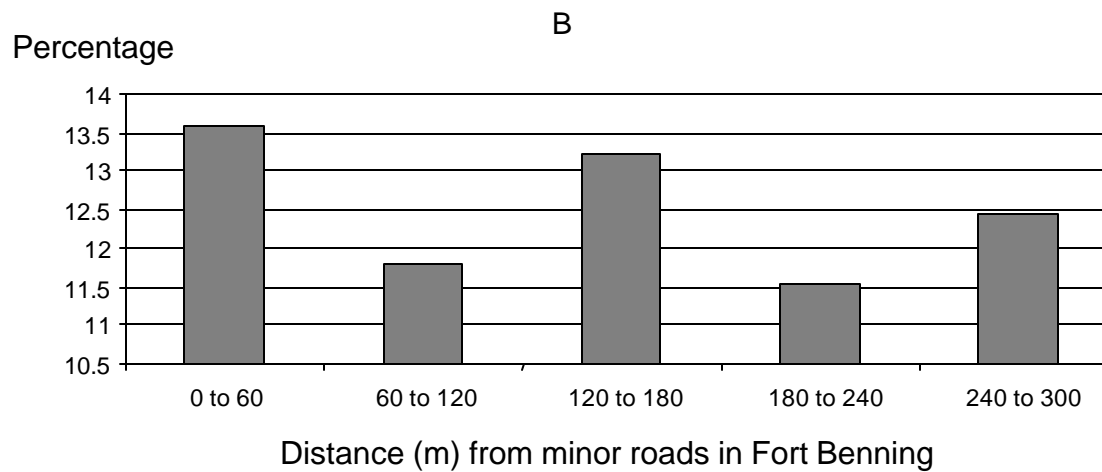
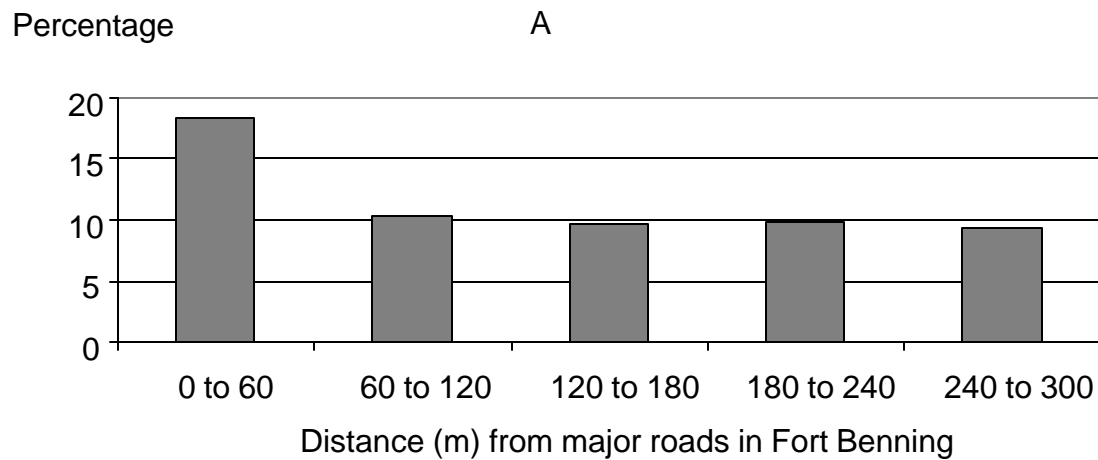




Figure 4. Results of the change detection from 1974 to 1999 performed for the land cover of buffered roads at Fort Benning for distances from (A) major roads (interstates, two- and four-lane highways), (B) minor paved roads, and (C) unpaved roads and tank trails. The percent conversion from forest to nonforest land-cover categories at different distances from the roads is plotted for the time period 1974 to 1991. The percentage indicates new changes for each buffer distance in comparison with smaller buffer areas.





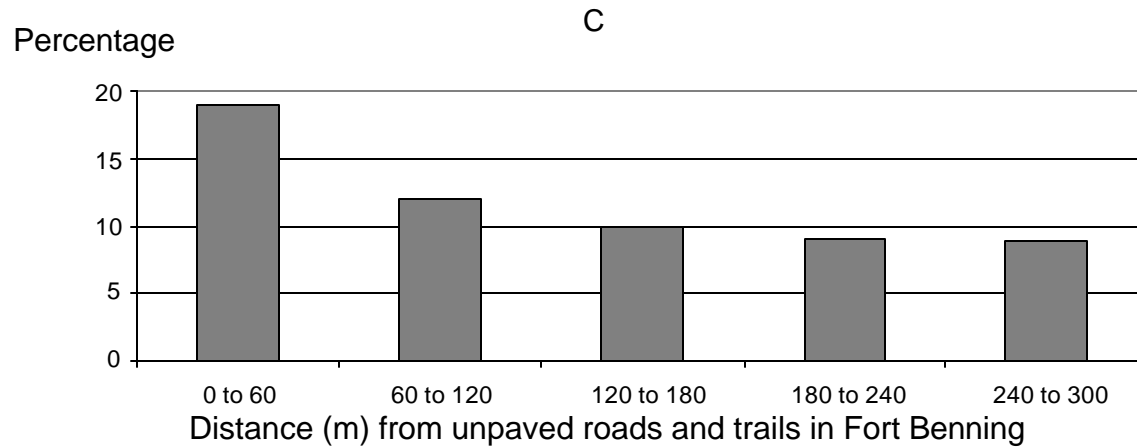
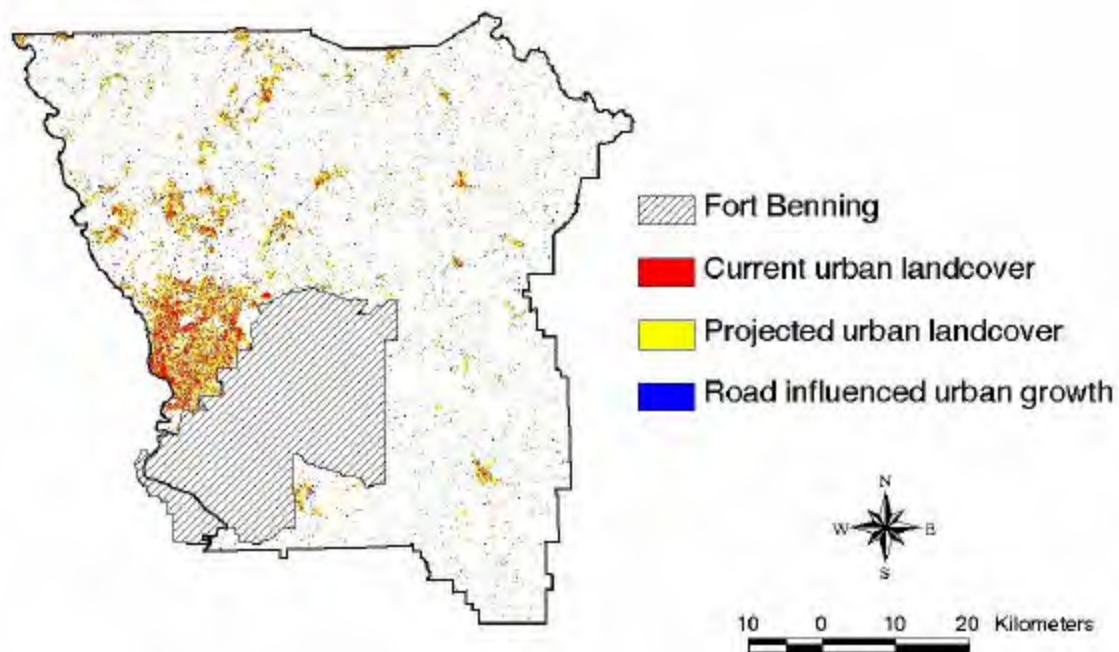


Figure 5. Map of the current and projected urban areas in the five-county area around Fort Benning and the small area of projected growth due solely to roads.



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**Conclusions of SEMP Research Project:**  
***Indicators of Ecological Change***  
**Principal Investigator: Virginia H. Dale**  
**March 2005**

**1. Landscape indicators**

Data collected for disparate purposes can be used to help develop an understanding of land-cover changes over time and are often necessary to further our knowledge of historic conditions on a given landscape. For the entire Fort Benning landscape, the values of landscape metrics for 1827 were very different from the values for recent decades. While the changes between 1827 and 1974 may be somewhat exaggerated due to data constraints, we can conclude that the nineteenth century landscape at Fort Benning was composed largely of uninterrupted pine forest with some deciduous forests found in riparian corridors and some open areas associated with Native American settlements. Land cover and land use in the 1970s were considerably different. Following decades of farming, military training activities had a pronounced effect upon the landscape. Heavy training activities resulted in areas of sparse land cover and bare ground. Interestingly, these areas have largely persisted on the landscape throughout the 1980s and 1990s. This result not only emphasizes the lasting footprint that military activities have on the landscape but also highlights the efforts made by management to confine heavy training exercises to certain sacrifice areas. Another interesting trend occurred in the 1990s. Pine forests have been on the rise as is reflected in both landscape composition and patch dynamics such as largest patch size, number of patches, and total edge. Management efforts at Fort Benning have focused upon managing for longleaf pine. These efforts appear to be decreasing hardwood invasion in favor of pine species in many areas on the installation.

Examining a suite of landscape metrics over time was useful for summarizing, describing, and assessing land-cover change at Fort Benning. The FRAGSTATS and ATtILA programs were relatively simple to use and provided information pertinent to understanding and managing the land. Therefore, we encourage resource managers to use landscape metrics to analyze changes in patterns of land cover over time to examine how human activities have affected an area.

Furthermore, work has already begun on obtaining and classifying historical aerial photography from the 1940s and 1950s. The characterization of additional time periods between 1827 and 1974 will be extremely useful in bridging the gap in our understanding of the landscape dynamics between the nineteenth century landscape and the established military base of the 1970s.

**2. Watershed indicators**

We found that a number of physical, hydrological, chemical, and biological characteristics of streams were good indicators of watershed-scale disturbance at FBMI. Stream channel organic variables (i.e., BPOM, CWD) were highly related to disturbance and thus were good indicators. Additionally, the degree of hydrologic flashiness (as quantified by 4-hour storm flow recession constants) and bed stability were good indicators of watershed-scale disturbance. Among the stream chemistry variables, the concentrations of total and inorganic suspended sediments during baseflow and storm



periods were excellent indicators of disturbance, increasing with increasing disturbance levels. In addition, baseflow concentrations of DOC and SRP were good disturbance indicators, declining with increasing disturbance levels. The magnitude of increases in SRP and possibly  $\text{NO}_3^-$  concentrations during storms also appeared to be good disturbance indicators. Among the biological variables, stream benthic macroinvertebrates also served as good indicators of watershed-scale disturbance. Traditional measures such as richness measures (e.g., number of EPT taxa and richness of Chironomidae) negatively corresponded with watershed disturbance; however, except for chironomid richness, all measures showed high variation among seasons and annually. A multimetric index previously designed for Georgia streams (GASCI) consistently indicated watershed disturbance and exhibited low seasonal and annual variation. Low diversity of fish precluded use of traditional measures (i.e., richness, diversity), however the proportional abundance of the two dominant populations (*P. euryzonus* and *S. thoreauianus*) were strongly but oppositely associated with disturbance, with *P. euryzonus* and *S. thoreauianus* being negatively and positively related to disturbance, respectively. Finally historic land use explained more variation in contemporary bed stability and longer-lived, low turnover taxa than contemporary land use suggesting a legacy effect on these stream measures. Prior to identification and use of potential indicators, we recommend that FBMI land managers consider land use history and the potential for legacy effects on contemporary conditions in streams.

### 3. Plot level indicators

We hypothesized that understory diversity and cover sampled from an anthropogenic disturbance gradient within the longleaf pine forests would reveal significant compositional and structural differences that occurred as a result of military training intensity. The confirmation of life form distribution and plant family cover as distinguishing features suggests that monitoring programs for longleaf pine forests should include understory vegetation as an ecological indicator. These metrics can serve as surrogate measures of disturbance to the longleaf pine system. Both life form distribution and plant family cover appear to be useful ways to group the large number of species which occur in the understory of these longleaf pine forests.

Indicators of disturbance that are used for resource management need to be easy to measure, sensitive to stresses, and predictable as to how they respond to stress. Selecting indicators for the understory of longleaf pine forests is complicated by the high species diversity. Field classification of understory plants according life form and family is relative straightforward compared to species identification. Both of these attributes are relatively easy and time efficient to measure and interpret. Thus we recommend that the suite of indicators used for monitoring longleaf pine ecosystems include these metrics.

Another goal of this project was to explore the possibility of using the soil microbial community as an ecological indicator signaling the degree of environmental degradation along a disturbance gradient. The analysis based on the soil PLFA was successful, reflected above-ground changes, and provided an index of the degree of land disturbance (traffic) the soil received. Both linear discriminant and non-linear ANN analysis were able to adequately classify the degree of disturbance. However, there were drawbacks when the ANN and linear discriminant models were used to predict stages of soil recovery in remediated transects. The linear discriminant model was shown to be a



fairly robust but perhaps coarse measure of remediative efforts, and the ANN was sufficiently sensitive to detect subtleties in recovery not detected with the linear discriminant analysis, but in current form could not be relied on to classify remediated samples. The inclusion of data reflecting remediation in these models could possibly make them capable of monitoring a more complete process of soil degradation and recovery. Soil microbial analysis provides indicators of ecological condition that respond in a short time to changes in land-use practices. Field collection requires minimal work, and samples can be shipped to commercial laboratories for analysis. Soil microbial condition is an important metric to gauge changing conditions on the land at Fort Benning.

#### **4. Overall summary**

These studies support our hypothesis that a suite of metric is useful for measuring changes in ecological conditions at Fort Benning. That suite should include landscape metrics of current and historical conditions, watershed indicators, and plot-level metrics (including both changes in vegetation and soil microbial biology). Together these indicators reveal information about changes at critical spatial and temporal. For specific management questions, the resource managers are urged to select from the suite of indicators analyzed in this research as well as those presented by other researchers for those indicators that best meet the criteria for the task.









## **Technology Transfer of Ecological Indicators:**

### **Overview of SERDP Ecosystem Management Project**

#### **“Indicators of Ecological Change”**

**February 2005**

Virginia H. Dale, Principal Investigator  
Oak Ridge National Laboratory

As the Department of Defense’s (DoD) Corporate Environmental R&D Program, the Strategic Environmental Research and Development Program (SERDP) has a commitment to focus on conservation technologies that can assist DoD in meeting its meeting mission needs. Because the current costs of environmental conservation challenges are significant and can even jeopardize mission activities, it makes sense to implement ecosystem management on the DoD installations. Ecosystem management requires the identification of ecological attributes that are indicative of critical processes of an ecosystem and that can be altered by management actions. Therefore, this effort is designed to develop indicators of ecological change at Fort Benning, Georgia, that should be applicable to other DoD installations.

Characterizing how resource use and management activities affect ecological conditions is necessary to document and understand ecological changes. Resource managers on military installations have the delicate task of balancing the need to train soldiers effectively with the need to maintain ecological integrity. Ecological indicators can play an important role in the management process by providing feedback on the impacts that training has on environmental characteristics, including soil chemistry, soil microbes, and vegetation.





## Participants



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- **Thomas Foster**, Anthropology Department, Pennsylvania State University — **Historical land cover**
- **Patrick Mulholland**, Environmental Sciences Division, Oak Ridge National Laboratory — **Aquatic ecology**
- **Lisa Olsen**, Environmental Sciences Division, Oak Ridge National Laboratory — **Geographic information and landscape analysis**
- **David White, Aaron Peacock, James Cantu, and Sarah McNaughton**, Center for Environmental Technology, University of Tennessee — **Soil microbiology**
- **Virginia Dale and Dan Druckenbrod**, Environmental Sciences Division, Oak Ridge National Laboratory — **Terrestrial and landscape indicators, integration**

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## Organization of Information

- Overview
- Landscape Indicators
- Watershed Indicators
  - Chemistry
  - Aquatic Organisms
- Plot Indicators
  - Vegetation
  - Microbes
- Conclusions and Discussion



## Overview: Challenges in the Development and Use of Ecological Indicators

### Outline:

- **How are indicators used to assess ecological condition?**
  - Definitions and purposes of indicators
  - Criteria for selecting indicators
  - Different perspectives on indicators
    - International
    - EPA
    - Other national approaches
- **Scientific challenges**
  - Scaling
  - Capturing complexity
  - Integration
  - Engaging resource managers & the public

This overview provides an introduction to indicators and to the in-depth discussion about particular indicators for Fort Benning.



## **How do we assess ecological condition using ecological indicators?**



Many groups have been developing methods and approaches to characterize ecological condition.



## Indicators are characteristics of the environment that, when measured, quantify the following\*

- **Magnitude of stress**
- **Habitat characteristics**
- **Degree of exposure to the stressor, or**
- **Degree of ecological response to the exposure**

\* Hunsaker and Carpenter. 1990. US EPA.

A challenge in using ecological indicators is determining which of the numerous measures of ecological systems best characterize the entire system but are simple enough to be effectively monitored and modeled. Ecological indicators quantify the magnitude of stress, degree of exposure to the stresses, or degree of ecological response to the exposure (Hunsaker and Carpenter 1990, Suter 1993) and are intended to offer a simple and efficient method to examine ecological composition, structure, and function of whole systems (Karr 1981) (e.g., see Table 1). The use of ecological indicators as a monitoring device relies on the assumption that the presence or absence of, and fluctuations in, these indicators reflect changes taking place at various levels in the ecological hierarchy (Noon et al. 1999).



## THE SPECTRUM OF ENVIRONMENTAL PERFORMANCE MEASURES

### EPA's Science Advisory Board Proposed Terminology

| Administrative Measures                                 | Stressor Measures                         | Exposure Measures                                      | Effects/Conditions Measures                                |
|---|---|--|--|
| e.g., number of TMDL's completed for metals             | e.g., concentration of metals in a stream | e.g., concentration of metals in stream biota          | e.g., composition and diversity of stream biotic community |
| <hr/>   |   |  |  |
| <i>Least directly related to Environmental Outcomes</i> |   | <i>Most directly related to Environmental Outcomes</i> |  |



**EPA Science Advisory Board**

**These effects/conditions measures are the focus of this effort.**

The Environmental Protection Agency considers four types of environmental performance measures: administrative, stressor, exposure, and effects/conditions.



## Purposes of Indicators\*



- Assessment
- Documenting trends
- Early warning
- Diagnostic
- Linkages



\* Cairns et al. 1993. Hydrobiologia 263:1-44.

Ecological indicators have several purposes (Cairns et al. 1993). They can be used to assess the condition of the environment or to monitor trends in condition over time. They can provide an early warning signal of changes in the environment, and they can be used to diagnose the cause of an environmental problem. The purpose influences the choice of ecological indicators. However, tradeoffs between desirable features, costs, and feasibility often determine the choice of indicators.



## Indicators should:

- Have clear meaning
- Occur well in advance of required action
  - Capture the complexities of the ecosystem
  - Be simple enough to be easily and routinely monitored

## Examples

- Canary in mines
- Red algae in lakes

Selection of effective indicators is key to the overall success of any monitoring program. In general, ecological indicators need to capture the complexities of the ecosystem yet remain simple enough to be easily and routinely monitored.

### An example of an ecological indicator

An example of an ecological indicator is the presence of the cyanobacteria *Oscillatoria rubescens* in lakes that are on the verge of extreme eutrophication. The role of this cyanobacteria as an indicator was first identified in Lake Washington (Edmondson and Lehman 1981; Edmondson 1991). In the first half of the 20th century, metropolitan Seattle discharged treated sewage high in phosphorus content into Lake Washington. By 1955, effluent contributed more than 50% of the total phosphorus input into the lake. Increased nutrient levels altered lake productivity and resulted in massive blooms of cyanobacteria that negatively affected fish populations and greatly reduced water clarity. Public attention was called to the problem, and the resulting reversal of this eutrophication process occurred when sewage was diverted from the lake and into Puget Sound. The resulting drop in nutrient additions eliminated algal blooms and increased water clarity. Now, *Oscillatoria rubescens* is used as an indicator of impending eutrophication worldwide. It satisfies three elements of an ecological indicator in that it is easily measured, it signifies an impending change in the ecosystem, and both the potential ecosystem change and the high level of the indicator can be averted by management action. (Unfortunately, Puget Sound still suffered even after advances were made in the sewage treatment system).



## Historical View of How Ecological Indicators have been used:\*

1. Evident readily detectable signs (e.g. foul air)
2. Early warning of environmental change
3. Ecological sustainable systems



\* Rapport, D. 1992. Pages 121-134 in Ecological Indicators. London: Elsevier Applied Science.

The earliest efforts to develop environmental indicators came from attempts to deal with severe degradation related to growing urban populations in the 17<sup>th</sup> and 18<sup>th</sup> centuries as signaled by foul smelling and discolored waters of rivers flowing through cities and local fish kills. Thus earliest measures were of water quality (e.g., color, oxygen content).

The second wave of interest in indicators focused on whole ecosystems and sought to find early warning signals of change.

Today the emphasis is on sustainable systems and ecological services.



## An Example of the Use of Indicators in Weather

- Signs used to predict impending weather (e.g., clouds)
- Simple rules
  - “Red sky at night, sailors delight; red sky at morning, sailors take warning.”
  - Cue cards (used by pilots)
- Military roots
- The Weather Channel provides an example of how indicators have been adopted by the public



The use of indicators for weather provides a good example of how indicators might be used and used in environmental sciences. At first, simple signs and rules were used to understand and communicate indicators of impending weather. The military took a strong interest and developed models and tools to help forecast weather. Today the Weather Channel provides round-the-clock information on weather systems and educates the public about terminology, conditions, and the importance of weather. The study of weather has become an integral part of most people's lives.



## Criteria for Selection of Ideal Set of Indicators\*

- Easy to measure
- Sensitive to stress
- Predictable
- Anticipatory
- Predict changes that can be averted by management actions

\*Dale and Beyer. 2001. Ecological Indicators 1:3-10.

• **Be easily measured.** The indicator should be straightforward and inexpensive to measure. The metric needs to be easy to understand, simple to apply, and provide information to managers and policymakers that is relevant, scientifically sound, easily documented, and cost-effective (Stork et al. 1997, Lorenz et al. 1999). Historically, canaries were carried into mines to warn workers of the presence of methane and other gases that can lead to an explosion. The death of a canary is an easily observed result of the presence of volatile gases. When a bird succumbed to toxic gas, it was a graphic signal for the miners to leave the area immediately.

• **Be sensitive to stresses of the system.** The ideal ecological indicator is responsive to stresses placed on the system by human actions while also having limited and documented sensitivity to natural variation (Karr 1991). While many indicators may respond to all changes in the system, the most useful indicator is one that displays high sensitivity to a particular stress, thereby serving as an early indicator of reduced system integrity. For example, the gopher tortoise (*Gopherus polyphemus*) that occurs at Fort Stewart, Georgia, is highly sensitive to soil disturbances, and their absence in otherwise suitable sites suggests past physical disturbances. (This interpretation of the tortoise's absence in sand hills areas can be made only when there are no other pressures on the tortoise, such as harvesting tortoise for food or gassing burrows to collect snakes.)

• **Respond to stress in a predictable manner.** The indicator response should be clear and predictable even if the indicator responds to the stress by a gradual change (such as the increase in density of *Oscillatoria rubescens* in polluted lakes). Ideally, there is some threshold level at which the response observed is lower than the level of concern of the impact.

• **Be anticipatory, that is, signify an impending change in key characteristics of the ecological system.** Change in the indicator should be measurable before substantial change in ecological system integrity occurs. For the canary in the coal mine example, the birds died at levels of toxic gases not quite sufficient to create an explosion or be toxic to humans.

• **Predict changes that can be averted by management actions.** The value of the indicator depends on its relationship to changes in management actions. For example, the presence of young longleaf pine (*Pinus palustris*) serves as a measure of the recurrence of fire at Eglin Air Force Base (AFB) on the Florida panhandle (McCay 1998). With fire suppression, the normally restricted distribution of sand pine (*P. clausa*) expanded from 2,400 ha to over 24,000 ha and young longleaf pine became rare. It has only been through the reintroduction of a regular fire regime at Eglin AFB that the historically dominant species, longleaf pine, has been reestablished. Today, the presence of young longleaf pine, which survive light fire, is a testament to the two- to three-year fire regimes that occur not only at Eglin AFB but also at Forts Benning and Stewart in Georgia.



## Criteria for Selection of Ideal Set of Indicators\* (Continued)

- Known response to natural disturbances, anthropogenic stresses, and changes over time
- Low variability in response
- Integrative
- Broadly applicable
- Consider spatial and temporal context

\*Dale and Beyer. 2001. Ecological Indicators 1:3-10.

• **Together with the full suite of indicators, provide a measure of coverage of the key gradients across the ecological systems (e.g., soils, vegetation types, temperature, space, time, etc.).** The full suite of indicators for a site should integrate across key environmental gradients. For example, no single indicator is applicable across all spatial scales of concern. Brooks et al. (1998) developed a suite of indicators for forested riparian ecosystems of Louisiana that behave predictably across scales and can be aggregated to provide an assessment of the entire system. We similarly propose a candidate suite of indicators for Fort Benning, Georgia, that together characterize the spatial scales of interest. Indicators can also be displayed along a temporal scale to represent the times of management interest. In a like manner, the ability of the suite of indicators to embody the diversity in soils and other environmental gradients at a site could be considered.

• **Have a known response to natural disturbances and changes over time.** The indicator should have a clear reaction to both natural disturbance and to anthropogenic stresses in the system. This criterion would pertain to conditions that have been extensively studied and have a clearly established pattern of response. Focal species are often the only types of species that have a large enough foundation of information to indicate long-term trends and responses to change. Landscape structure can also respond to human disturbances versus natural disturbances in a predictable manner (Krummel et al. 1987). Thus, landscape metrics can serve as useful indicators of change (Dale and Pearson 1997).

• **Have low variability in response.** Indicators that have a small range in response to particular ecosystem stresses allow for changes in the response value to be distinguished from background conditions. As a counter example, seabirds were a poor indicator of the ecological cost of the 1989 Exxon Valdez oil spill and of the benefit of subsequent steam cleaning. More than 30,000 oiled bird carcasses were retrieved following the spill, but because of the high variability inherent in seabird population changes, the population dynamics of birds in the spill areas are difficult to distinguish from the population dynamics of birds in the nonspill sites (Wiens 1996).



## International Approach: Santiago Declaration

### Indicators Provide\*

- Key terms that can be used in future international discussions
- A framework against which countries can evaluate their policies
- A mechanism for rationalizing and evaluating international aid and technical cooperation programs
- Means of resolving difficulties in international discussions on trade in products



\* Brand DG. 1997. **BIOMASS & BIOENERGY** 13: 247-253.

At their Sixth Meeting of the Montréal Process Working Group in Santiago, Chile, February 1995, ten nations agreed to a comprehensive set of criteria and indicators for forest conservation and sustainable management. This statement of endorsement is referred to as the "[Santiago Declaration](#)". They were joined in October 1995 by two more nations, Argentina and Uruguay, completing the current group of twelve member countries.

(see: [http://www.mpci.org/santiago\\_e.html](http://www.mpci.org/santiago_e.html))

The Montréal Process Criteria and Indicators are intended to be applied, at the national level, to all the forests of a country, across all types of land ownership. They consider sustainable forest management in a holistic way, taking into account all forest goods, values and services. By endorsing these Criteria and Indicators, each participating country has made a commitment to work toward the sustainable management of its forests.



## International Approach: Center for International Forestry (CIFOR)\*

- **Related indicators to stressor**
- **Hierarchical approach**
  - Landscape pattern
  - Habitat structure
  - Community guild structure
  - Taxonomic richness & composition
  - Population structure
- PLUS**
  - Ecosystem processes
  - Water quality & quantity

| Stressor  | Indicators         |                               |                           |                             |
|-----------|--------------------|-------------------------------|---------------------------|-----------------------------|
|           | Cause/<br>impact   | Habitat<br>fragmen-<br>tation | Air<br>quality<br>decline | Water<br>quality<br>decline |
|           | Land use<br>change | 44554                         | .667                      | .0334                       |
|           | Land<br>management | 343490                        | .770                      | .00334                      |
| Pollution | 45490              | .444                          | .9944                     |                             |

\* Stork et al. 1997. CIFOR paper #17.

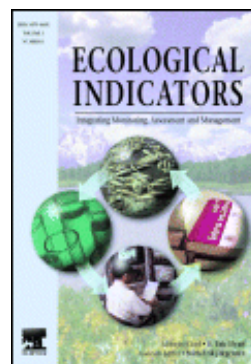
The Center for International Forestry (CIFOR) has developed a comprehensive set of criteria and indicators for sustainable forest management based on CIFOR's research. This research was conducted by interdisciplinary teams of experts in large-scale natural forests managed for commercial timber production in Indonesia, Côte d'Ivoire, Brazil and Cameroon, with additional sites in Germany, Austria and USA. It is envisaged that this template would be used primarily for tropical natural forests managed for commercial purposes.

(See: <http://www.cifor.cgiar.org/acm/methods/toolbox2.html>)



## Indicators are the Focus of Much Attention

- **International**
  - Santiago Declaration
  - Center for International Forestry (CIFOR)
  - New journal: *Ecological Indicators*
- **National**
  - Heinz report: *The State of the Nation's Ecosystems*
  - National Academy report: *Ecological Indicators for the Nation*
  - EPA's *Report on the Environment*
  - *Wildlife Society's Performance Measures*
  - *EPA's Indicators of Watershed Health*



**But many efforts do not focus on local level of resource management**

**Heinz report: The State of the Nation's Ecosystems --**  
<http://www.heinzctr.org/ecosystems/index.htm>

**National Academy report: Ecological Indicators for the Nation --**  
<http://books.nap.edu/catalog/9720.html>

**EPA's Report on the Environment --**  
<http://www.epa.gov/indicators/roe/index.htm>

**Wildlife Society's Performance Measures --**  
<http://www.wildlife.org/wg/biodiversity/index.cfm?tname=conference2004>

**EPA's Indicators of Watershed Health --**  
<http://www.epa.gov/bioindicators/>

***Ecological Indicators* (the journal) --**  
[http://www.elsevier.com/wps/find/journaldescription.cws\\_home/621241/description#description](http://www.elsevier.com/wps/find/journaldescription.cws_home/621241/description#description)



## National reports demonstrate the many gaps in information about indicators

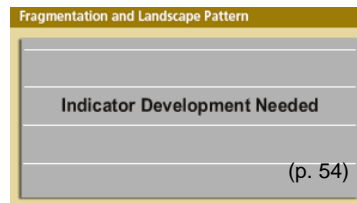
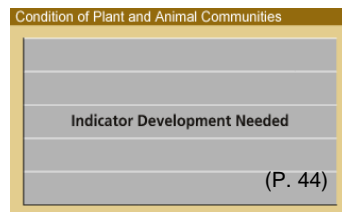
### National Academy report:

"Only rough estimates of the value to society of ecological goods and services are available" (p. 26)

### EPA's Report on the Environment:

- "Further exploration of the relationship between measurements used for assessments and measurements used for diagnosis of causal factors ... is needed" (p. v)
- "The availability of data were too limited in place or time to describe national trends, or even to provide a national snapshot of conditions" (p. vi)

**Heinz Report:** illustrates data gaps and show where work is needed



The efforts to identify ecological indicators that are available for the entire nation show that there are many gaps in knowledge and data.



## Currently only a few indicators have data available to describe subnational trends

- **Data must be**
  - Complete
  - Current
  - Reliable
  - Available regionally or statewide
- **Opportunities exist**
  - To develop data bases
  - To design indicator analysis
  - To better understand local conditions
- **Challenge**
  - To design experiments to test indicators



It is a challenge to identify indicators that are appropriate at local and regional levels. Our project designed and implemented sampling protocols and experiment to develop and test such indicators.



## Definitions must be clear and measures repeatable

- Various definitions of rangelands and forests can cause a 10 to 15% variance in area\*
  - What is a tree?
  - How much crown cover is needed for a forest?



\* Goebel et al. 1998 (Oregon Demonstration Project)

A critical first step in developing indicators is to clarify language so that the terms are understood by all parties and therefore useful in accomplishing goals.

There is the semantic aspect to this change. For example, foresters have long argued about what constitutes a forest (is it denoted by basal area, tree density, canopy cover, or something else). Clearly, when all of the trees are gone, the area is no longer under a forest. Yet, the land use can still be in forestry if the area is treated as plantations or left to succeed to trees.



## Concerns About the Use of Indicators: Monitoring and Management Programs Often\*

- Depend on a small number of indicators
- Fail to consider the full complexity of the ecological system
- Have vague long-term goals and objectives
- Lack a scientific rigor
- Fail to use defined protocols for identifying ecological indicators



\* Cairns et al. 1993. Hydrobiologia 263:1-44.

• **Management and monitoring programs often depend on a small number of indicators and, as a consequence, fail to consider the full complexity of the ecological system.** By selecting only one or a few indicators, the focus of the ecological management program becomes narrow, and an oversimplified understanding of the spatial and temporal interactions is created. This simplification often leads to poorly informed management decisions. Indicators should be selected from multiple levels in the ecological hierarchy in order to effectively monitor the multiple levels of complexity within an ecological system.

• **Choice of ecological indicators is often confounded by management programs that have vague management goals and objectives.** Unclear or ambivalent goals and objectives can lead to “the wrong variables being measured in the wrong place at the wrong time with poor precision or reliability” (Noss and Cooperrider 1994). Primary goals and objectives should be determined early in the process in order to focus management. Ecological indicators can then be selected from system characteristics that most closely relate to those management concerns.

• **Management and monitoring programs often lack scientific rigor because of their failure to use a defined protocol for identifying ecological indicators.** Lack of a procedure for selecting ecological indicators makes it difficult to validate the information provided by those indicators. Until a standard method is established for selecting and using indicators, interpretation of their change remains speculative (Noss 1999). The creation and use of a standard procedure for the selection of ecological indicators allow repeatability, avoid bias, and impose discipline upon the selection process, ensuring that the selection of ecological indicators encompasses management concerns (Slocombe 1998, Belnap 1998).



## Scientific challenges

- **Scaling**
- **Capturing complexity**
- **Engaging resource managers & the public**

The rest of this overview discuss the three key challenges of

- Scaling
- Capturing complexity
- Engaging resource managers & the public



## Scaling Challenge:

**Indicators at one scale may reveal  
information about another level.**

The challenges of scaling is determining the appropriate scale to obtain measure and the appropriate level at which to convey the information to decision makers.



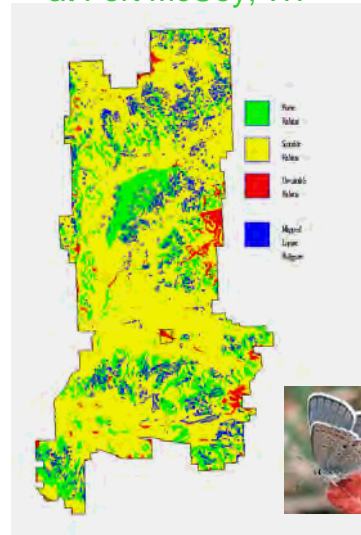
## Different Spatial Scales

### DoD Installations in US



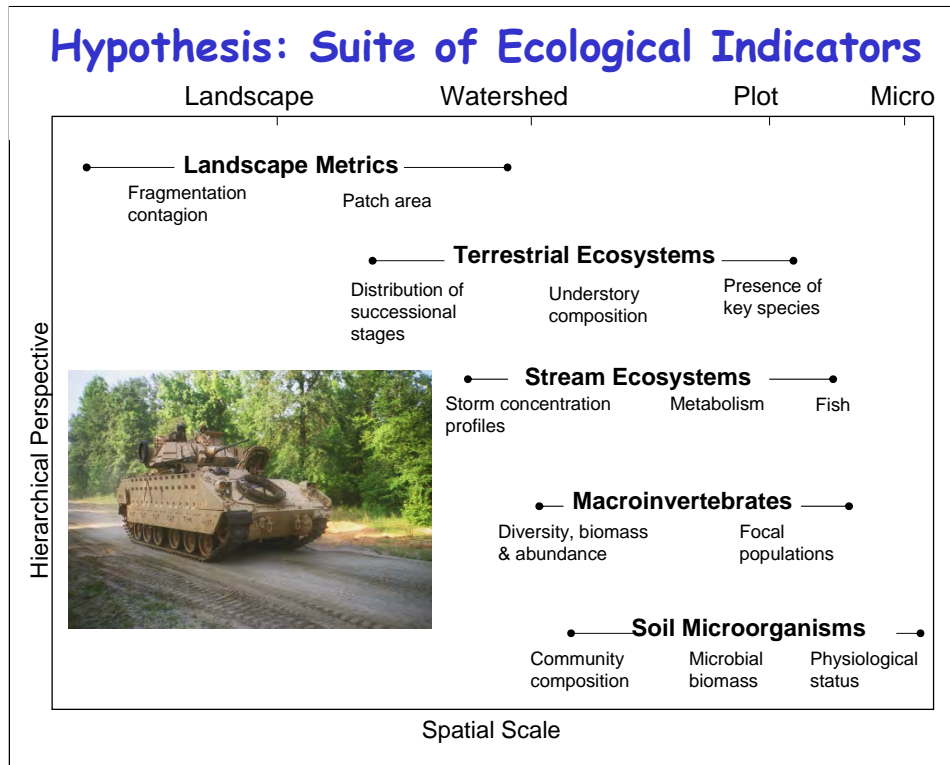
Plot studies at Fort Benning, GA

### Habitat for Karner Blue Butterfly at Fort McCoy, WI



Military installations obtain information at different spatial scales and must use information at different spatial resolutions. For example, it is important to understand characteristics of all the military installations in the US (e.g., that they contain one of the highest number of federally listed species per land area of any of the federal land owners). At an installation scale of perspective, it is critical to know where potential habitat occurs for rare species (e.g., see the map of potential habitat for Karner blue butterfly at Fort McCoy, WI). At the local level, resource managers need to know where a rare species occurs.





At Fort Benning, we examine the hypothesis that indicators about the features at several scales of resolution are all necessary to explain changes in ecological conditions associated with land cover and use activities.

Dale, V H., Mulholland, P., Olsen, L. M., Feminella, J., Maloney, K., White, D. C., Peacock, A., and Foster, T., “**Selecting a Suite of Ecological Indicators for Resource Management**,” *Landscape Ecology and Wildlife Habitat Evaluation: Critical Information for Ecological Risk Assessment, Land-Use Management Activities and Biodiversity Enhancement Practices*, ASTM STP 11813, L. A. Kapustka, H. Gilbraith, M. Luxon, and G. R. Biddinger, Eds., ASTM International, West Conshohocken, PA, 2004.



## Some Candidate Indicators for Plot Scale

- **Vegetation**
  - Tree density & diversity
  - Understory cover & diversity
  - Cover by life form
- **Soils**
  - Depth of A horizon (top layer of soil)
  - Surface soil carbon concentration
  - Surface soil bulk density
- **Soil fungi**



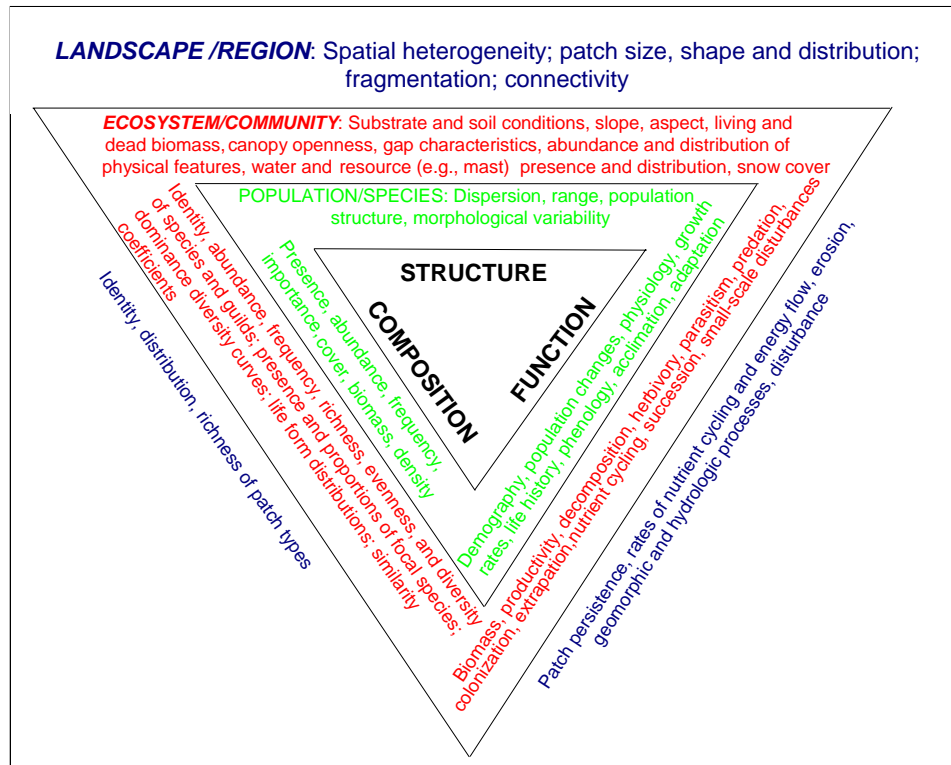
The objective of this study was to identify indicators that signal ecological change in intensely and lightly used ecological systems (all Fort Benning has had some anthropogenic changes). Because the intent was that these indicators become a part of the ongoing monitoring system at the installation, the indicators should be feasible for the installation staff to measure and interpret. While the focus is on Fort Benning, the goal is to develop an approach to identify indicators that would be useful to a diversity of military installations and other land ownerships (in some cases the actual indicators may be adopted). Because some environmental impacts may be long-term or may occur after a lag time, early indications of both current and future change need to be identified. The intent of this identification of indicators is to improve managers' ability to manage activities that are likely to be damaging and to prevent long-term, negative effects. Therefore, we examined a suite of variables needed to measure changes in ecological conditions. Hald et al. (2003) found that a combination of indicators was preferable when used to select grassland sites for successful botanical restoration. The suite that we considered for military land use at the plot level included measures of soil properties, soil microorganisms as a measure of below-ground integrity of the ecosystem, and terrestrial vegetation.



## Challenge: Capturing the complexity

As the complexity of environmental problems becomes recognized, it also becomes clear that information from only one sector of the ecological system may not be sufficient to address environmental concerns. The concept of ecological sustainability arose, in part, to capture the full complexity of features necessary to maintain the services of the Earth. For example, soil quality indices (Andrews and Carroll 2001) have been devised to combine various soil data in order to monitor the sustainability of agroecosystems. Many indicators of sustainability involve information about ecological, social, and economic aspects of life (e.g., Zhen and Routray 2003). Yet devising ways to monitor changes within the ecological system also requires a diverse approach.





The ecological hierarchy includes the functional, compositional, and structural elements that, when combined, define the ecological system and provide a means to select a suite of indicators representative of the key characteristics of the system. All ecological systems have elements of composition and structure that arise through ecological processes. The characteristic conditions depend on sustaining key ecological functions which, in turn, produce additional compositional and structural elements. If the linkages between underlying processes and composition and structural elements are broken, then sustainability and integrity are jeopardized and restoration may be difficult and complex.

Ideally the suite of indicators should represent key information about structure, function, and composition. The complexity of Figure 1 only hints at the intricacy of the ecological system on which it is based. The series of nested triangles in the figure are meant to suggest that knowledge of one part of the triangle may provide information to the other aspects of the system. For example, often it is easier to measure structural features that can convey information about the composition or functioning of the system than to measure composition or function (Lindenmayer et al. 2000). Sometimes measures from one scale can provide information relevant to another scale. For example, the size of the largest patch of a habitat often restricts the species or trophic levels of animals that are able to be supported based solely on their minimal territory size (Dale et al. 1994). Even so, it is often difficult to know how large an area or how long to monitor (Dawe et al. 2000). The ecological system can be viewed as a moving target (Walters and Holling 1990) with many system variables changing slowly and not stabilizing for a long time. Thus the diagram can be used in the selection of indicators for some aspects of the ecological hierarchy are easier to measure than other, and can be correlated to other aspects of the diagram.



# Challenge:

## Engaging resource managers & the public



## The challenge is developing indicators that provide managers with information on how resource use & management can coexist

### Example:



Characterizing how resource use and management activities affect ecological conditions is necessary to document and understand ecological changes. Resource managers on military installations have the delicate task of balancing the need to train soldiers effectively with the need to maintain ecological integrity. Ecological indicators can play an important role in the management process by providing feedback on the impacts that training has on environmental characteristics, including soil chemistry, soil microbes, and vegetation.



## Asked how Fort Benning resource managers might use indicators

### Their responses included:

- Planning budgets
- Provide a “heads up” regarding compliance
  - Heading toward non-compliance?
- Signal whether on right path toward achieving longer term goals
- Signal whether on right path to achieve shorter term objectives
- Suggest need for targeted research
  - The “holy cow” scenario



Photo: Fort Bragg

It was useful to sit down with the Fort Benning resource managers in the same room and facilitate discussions of how indicators might be used and useful to their daily work and programs.



## Measures of practical utility suggested by Fort Benning resource managers

- Provide feedback on whether current ecological conditions are consistent with achieving goals and objectives
- Indicator values are meaningful—quantifiable and able to signal “red flags”
- Help resource managers anticipate potential noncompliance
- Maximize the ratio of sampling effort exerted to information yielded (“biggest bang for buck”)
  - Sampling design, effort, & analyses should be proportionate to need
  - Sampling measurement should be cost-effective\*
  - Indicator should be comprehensive\*
    - Provide information about a large area, more than one resource, etc.
- “Cheaper and broadly applicable is better, but more expensive and narrowly applicable might be ok”
  - If associated with*
    - critical training needs
    - Endangered Species Act
    - isolated populations (“lucrative targets”)

The Fort Benning resource managers were most interested in the practical utility of indicators.



## Indicators are being placed in context of management needs



As suggested by the diagram above, needs for environmental management at Fort Benning are quite diverse.



**To engage resource managers & the public, indicators must deal with values. Yet values can be quite diverse.**

**Example: The Grand Canyon**

Sports & Fishing



Power



Endangered Species



Cultural Artifacts

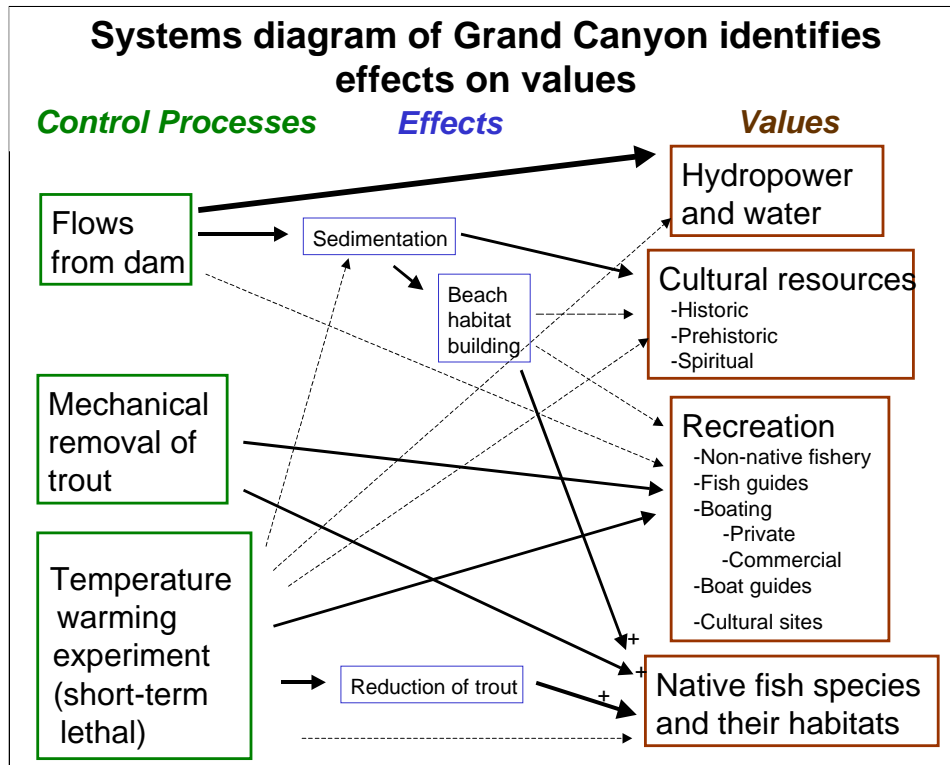


Boating: guides and tourists



Experiences with adaptive management at Grand Canyon show that there are a diversity of stakeholders, and each has different goals for resource management.





This diagram shows how different control processes for the Grand Canyon can affect the values of stakeholders. The multiple effects and strength of impact (as indicated by heaviness of the line) were important to understanding the ramifications of control actions.

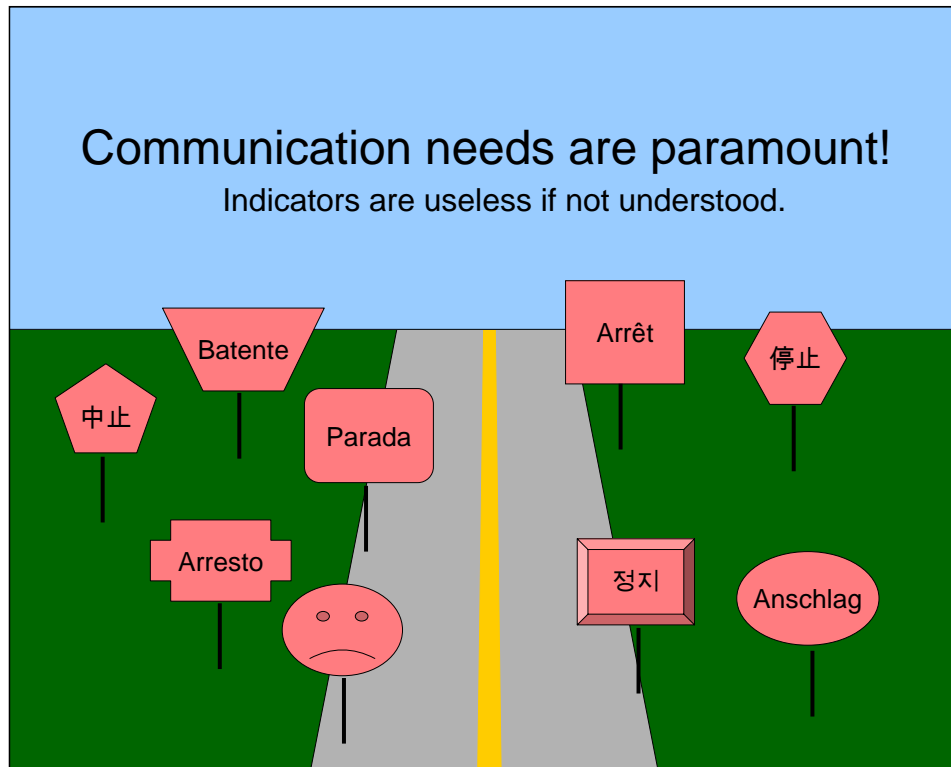


## Our focus is on:

- **Identifying indicators of ecological impacts of prior resource use or management.**
- **Determining how these indicators can be an integral part of the monitoring and management program**
- **Developing broadly applicable procedures**

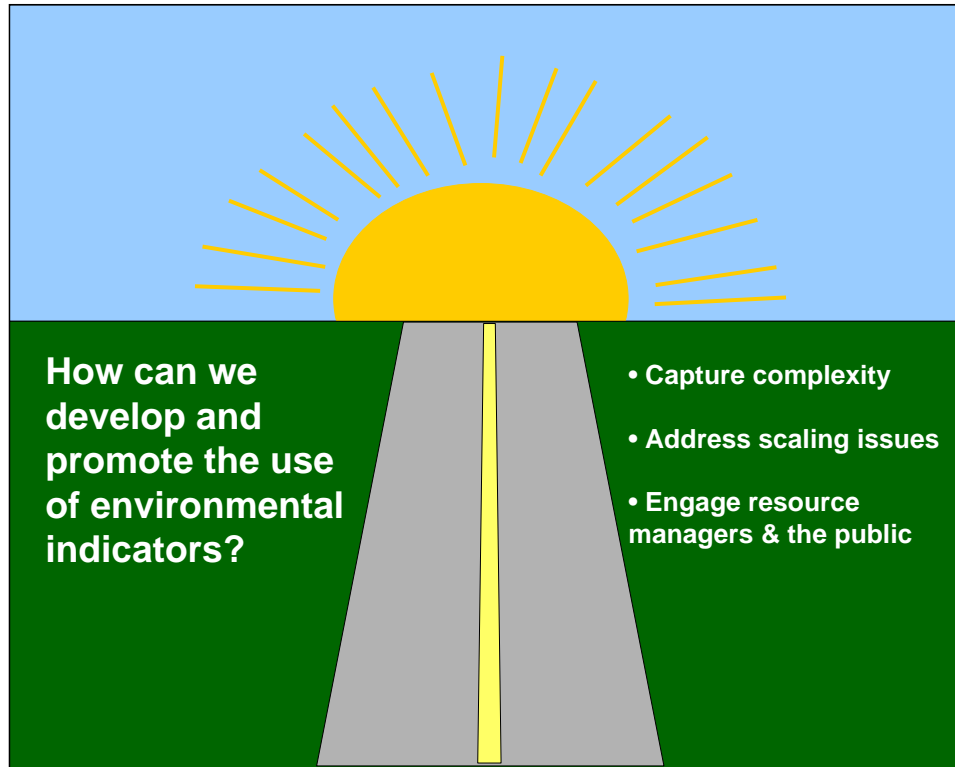
Our indicator research project at Fort Benning has three main foci defined above.





Warning signs, such as stop signs, cannot be understood if the language is unknown to those interpreting them.

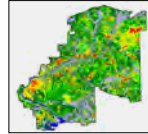




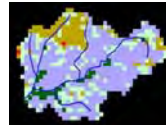


## The following sections focus on indicators at different scales

■ Landscape



■ Watershed



■ Plot



Our goal was to develop valid and repeatable indicators of impacts of training on understory vegetation, soils, and soil microbes in sites suitable for longleaf pine forests. Thus the study focused on maneuver training areas and does not include firing ranges, ordnance impact areas, or cantonment areas.

The rest of the briefing will focus on indicators at there scales of resolution:

- Landscape (including all areas of Fort Benning)
- Watershed
- Plot



## Technology Transfer of Ecological Indicators:

### Landscape Indicators

Virginia H. Dale and Lisa Mai Olsen  
Oak Ridge National Laboratory  
and  
Thomas Foster  
Panamerican Consultants, Inc

**OVERVIEW:** This research examined landscape indicators that signal ecological change in both intensely used and lightly used lands at the U.S. military installation, Fort Benning, Georgia. Changes in patterns of land cover through time affect the ecological system. Landscape patterns, therefore, are important indicators of land-use impacts, past and present, upon the landscape. This analysis of landscape pattern began with a landscape characterization based on witness tree data from 1827 and the 1830s and remotely sensed data from 1974, 1983, 1991, and 1999. The data from the early 1800s, although coarse, were useful in characterizing the historical range of variability in ecological conditions for the area.

The steps for the analysis involved the creation of a land-cover database and a time series of land cover maps, computation of landscape metrics, and evaluation of changes in those metrics over time as evidenced in the land-cover maps. An examination of land-cover class and landscape metrics, computed from the maps, indicated that a suite of metrics adequately describes the changing landscape at Fort Benning, Georgia. The most appropriate metrics were percent cover, total edge (km), number of patches, descriptors of patch area, nearest neighbor distance, the mean perimeter-to-area ratio, shape range, and clumpiness. Identification of such ecological indicators is an important component of building an effective environmental monitoring system.

Olsen, L.M., Dale, V.H. and H.T. Foster. In press. Landscape patterns as indicators of ecological change at Fort Benning, GA. **Land Use and Urban Planning**.



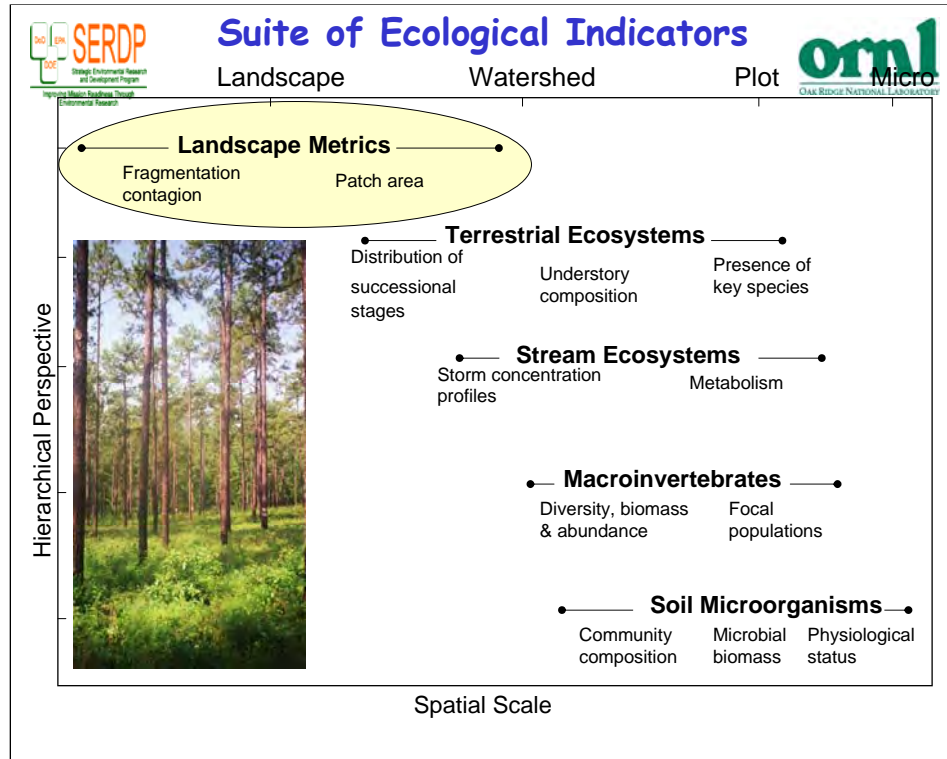
- **To identify broad-scale indicators of pattern that signal ecological change in intensely versus lightly used ecological systems.**
- **To ensure that these indicators are feasible for the installation staff to measure and interpret and thus can become a part of the ongoing monitoring system at the installation.**



Landscape metrics can capture changes in pattern, be implemented along a variety of spatial scales, and be useful indicators of land-cover changes due to prior land use and management. Landscape metrics should be useful to any landowner or manager interested in knowing whether the landscape pattern is changing and how. Such metrics could be particularly useful to the U.S. Department of Defense (DoD), which has taken a proactive approach to land management. A major component of DoD's mission is to provide adequate lands for military training and operations.

Military lands are subject to the same environmental responsibilities and regulations as other federal lands, and they contain a high density of federally and state listed species (Leslie et al., 1996), which by law require protection. Given the nature of military land use, military testing and training may degrade or fragment critical habitats and put species at risk. Vegetation and habitat play a crucial role in military training by providing concealment and an element of realism in training exercises. By harming the vegetation and habitat, military testing and training may degrade the very landscape characteristics that are necessary for thorough training and ultimately compromise the ability of a military installation to fulfill its mission, either by jeopardizing realism or by violating the provisions of the Endangered Species Act.





Together the suite of indicators should represent the range of ecological conditions in the ecological system, serve as signals of environmental change, and be simple enough to allow cost-effective monitoring and modeling (Hunsaker and Carpenter, 1990; Kelly and Harwell, 1990; Noss, 1990; Cairns et al., 1993; Dale and Beyeler, 2001). Landscape metrics that include quantifiable measures of landscape fragmentation have been developed to capture important aspects of landscape pattern in a few numbers (O'Neill et al., 1988; Riitters et al., 1995; Turner et al., 2001).



**Land degradation and habitat fragmentation resulting from military land use may compromise the ability of a base to fulfill its mission.**

- Importance of vegetation to military training
  - Concealment
  - Realism
- Environmental responsibilities: endangered species



As part of DoD's proactive approach to land management, environmental monitoring and management plans are being developed to assist military installations in balancing their training requirements and environmental responsibilities in both the short term and long term (e.g., Diersing et al., 1992). The immediate goal of our research was to identify and map trends in land-cover change that have occurred since 1827 at Fort Benning, Georgia, and to develop techniques to measure those trends. This analysis is a multi-step process beginning with the creation of land-cover maps for different time periods and culminating with the computation, summarization, and evaluation of landscape metrics.

Environmental management issues at Fort Benning focus on ways to retain or promote pine forests that support the federally endangered red cockaded woodpecker (*Picoides borealis*) and other habitats important to rare species (Addington, 2004), as well as environmental effects of sediment flux (largely coming from bare areas) (Fehmi et al., 2004). The resource managers are looking for ways to better monitor effects of their efforts to protect habitats of interest and to document impacts of military use on the environment (Addington, 2004). This study focused on broad-scale indicators of landscape change to support monitoring and trend assessment.



## Changes in Landscape Pattern are Important to quantify for:

- **Environmental monitoring and management**
- **Balancing training requirements and environmental responsibilities**
- **Determining impacts to landscape**



Specifically, we examined trends in landscape metrics that relate to changes in ecological conditions over time such as changes in vegetation type and pattern. This analysis identified landscape metrics that are useful indicators of change at Fort Benning. An indicator should adequately characterize an aspect of the system and be able to be implemented for management purposes (Dale and Beyeler, 2001). Although Fort Benning was the focal site of this project, the ultimate goal was to develop an approach for identifying landscape indicators that would be useful for a diversity of locations and management types. This wider effort is ongoing, and the results published here include a description of the methods and tools used to create the land-cover time series and to compute landscape metrics.



## Goals of this Research

- **To identify and map land cover trends that have occurred in recent human history at Fort Benning.**
  - Data collection
  - Development of land cover maps
- **To develop techniques to measure changes on the landscape.**
  - Computation, summarization, and evaluation of landscape metrics

As part of DoD's proactive approach to land management, environmental monitoring and management plans are being developed to assist military installations in balancing their training requirements and environmental responsibilities in both the short term and long term (e.g., Diersing et al., 1992). The immediate goal of our research was to identify and map trends in land-cover change that have occurred since 1827 at Fort Benning, Georgia, and to develop techniques to measure those trends. This analysis is a multi-step process beginning with the creation of land-cover maps for different time periods and culminating with the computation, summarization, and evaluation of landscape metrics.





## Technical Approach: Landscape Studies



- **Explore data availability**
- **Develop comparable data sets over time**
- **Analyze changes in pattern and distribution of land cover over time**
- **Determine which landscape metrics are most useful for indicating change**



- **Quantifiable measures of landscape fragmentation.**
  - Ex: vegetation type and pattern
- **Correlate with land use and ecological process**
- **Which metrics are useful *indicators* of change at Fort Benning?**
  - characterizes the system
  - easily measured or monitored by management
  - cost-effective
  - clear interpretation

Landscape metrics include quantifiable measures of landscape fragmentation that have been developed to capture important aspects of landscape pattern in a few numbers (O'Neill et al., 1988; Riitters et al., 1995; Turner et al., 2001). These numbers can often be correlated with land-use change and ecological processes. By using such metrics to examine and quantify landscape patterns through time, researchers may determine the long-term impacts of previous land use (e.g., Bürgi, 1999; Griffith et al., in press). Landscape metrics can thus capture changes in pattern, be implemented along a variety of spatial scales, and be useful indicators of land-cover changes due to prior land use and management.



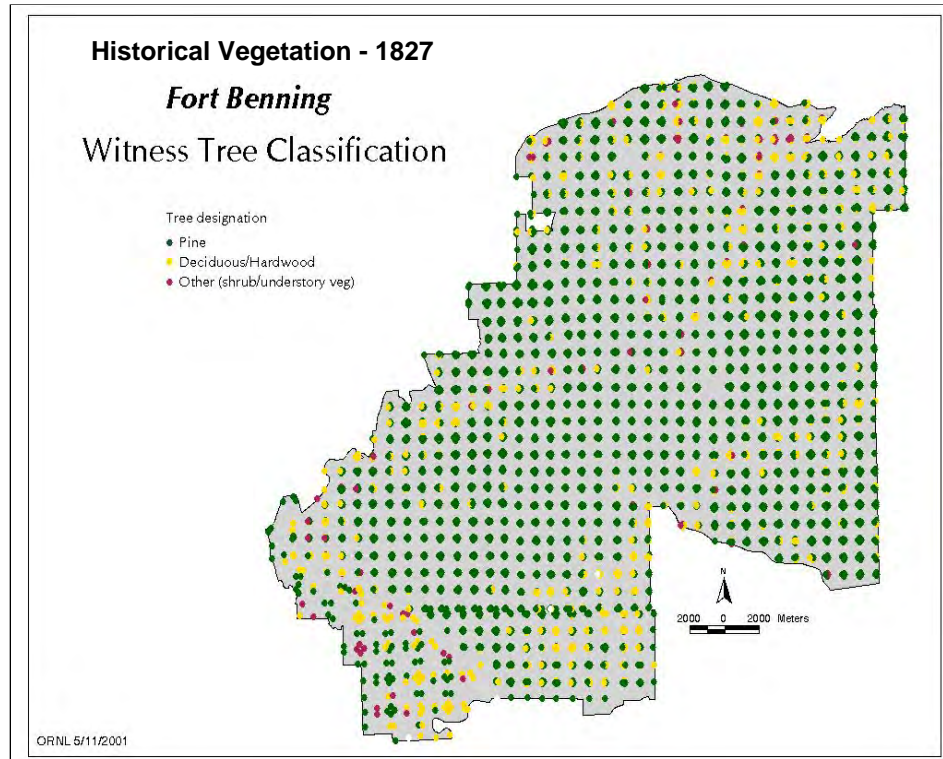
## Production of 1827 map of vegetation for Fort Benning

- **Witness tree data**
- **Tree classification**
- **Interpolation of surface**
- **Estimation of cleared areas -  
large settlements/towns**

To establish a baseline for the study, our analysis relied on witness tree data from land surveys conducted in 1827 and the 1830s. The historic land-survey data were obtained in the form of surveyors' notes and maps from the Georgia Department of Archives and History in Atlanta, from the Alabama Department of Archives and History in Montgomery, and from the Bureau of Land Management in Springfield, Virginia. Because the method of data preparation and integration can affect the results (Mladenoff et al., 2002; Petit and Lambin, 2002,), we describe our techniques in detail as well as relate the origin of the data.

The historical land survey maps were created in the 1800s, when the U.S. government surveyed the country for the purpose of subdividing and selling land. The surveyors partitioned federal lands into square townships that measured six miles on each side (1553 ha). Townships were further subdivided into sections of one square mile (258 ha). Surveyors marked the corners of each section and defined section boundaries by recording witness trees that were in close proximity to the boundary markers. In the 1830s, when the federal government conducted land surveys in the Alabama portion of the area that would become Fort Benning, corner and witness trees were recorded (Foster, 2001; Black et al., 2002). Earlier, when the Georgia portion was surveyed in 1827, the land was divided into roughly equal lots within districts. Districts measured approximately 2330 ha, and lots were 82 ha. The lots were then issued by a lottery. As a part of the land distribution, the Surveyor General surveyed the land, noting the location and species of the corner tree and four witness trees at each corner that marked the boundary of each lot. Unlike in other surveys, the surveyor did not routinely include information regarding bearing or distance from corner, diameter at breast height (DBH), or other indications of tree age or size. Nor did the surveyor include notes describing landscape or understory vegetation.





The land surveys as a whole show great variability in the information that was recorded about witness trees, largely as a result of personal biases of the individual surveyors and differences in their survey methods. Some surveyors recorded only common tree names (which varied greatly), and some reported trees only to the genus level. Others held biases in the species and sizes of trees selected as witness trees. It is often assumed that surveyors were biased towards longer living or larger trees when selecting witness trees (Black et al., 2002). In addition, some surveyors may have exaggerated the amounts of valuable timber species so that land values would be elevated. In spite of these problems, it is largely agreed that "witness tree data is the largest, most systematic, and most accurate form of data available for the pre-European settlement forests" (Bourdo, 1956; Whitney, 1994; Black et al., 2002).

The historical land survey maps and field notes were used to create a digital Geographic Information Systems (GIS) model of the forests covering the Fort Benning area. The model represents the forests in a Native American agricultural environment prior to extensive European settlement. As such, the model provides baseline conditions for the area currently occupied by the installation. The data extraction process involved georeferencing the historical survey maps, digitizing the location of the trees from the maps, and extracting the attribute species from the maps and survey notes. The historic maps for Georgia were georeferenced to modern U.S. Geological Survey (USGS) maps by using the Fort Benning Final Project and Acquisition Map EM 405-1-2-00 (U.S. Army Corps of Engineers, 1948) because it contained the survey boundaries from the historic maps. Modern aerial photos were occasionally used. The maps were digitized with a GTCO Accutab TM 24" x 36" (+/- 0.005" certified absolute accuracy). Data feature points were digitized as points in vector format. Trees were represented as points on the survey maps and in the digital GIS layer. The digitizing and GIS analysis were performed in Arcview 3.2a™ and Arc/Info 7.21™ (ESRI, Redlands, CA).

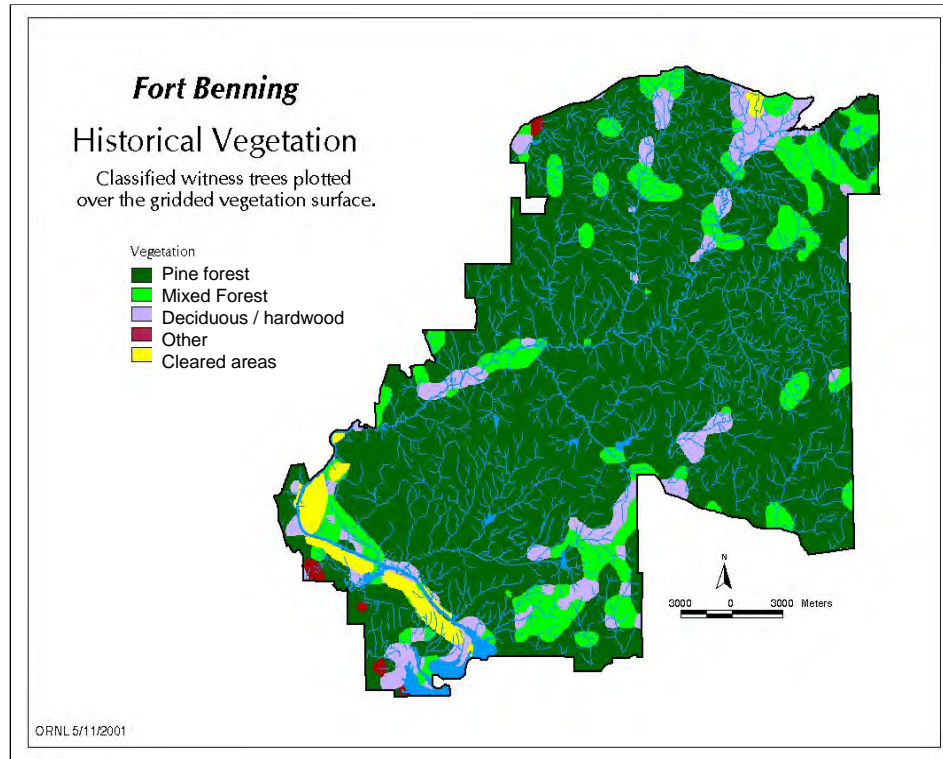
Common names of trees from the historic survey maps were assigned scientific names based upon Godfrey (1988). When ambiguous species were encountered, physiographic and habitat associations were used to clarify species (Black et al., 2002). Pine (*Pinus*) was recorded only at the genus level. Many of the corners are labeled only as "stake" or "post" on the plat maps to indicate how the lot corners were marked by the surveyor. In many instances, the species of the post is noted. After consultation with Mark Cowell (Department of Geography and Geology, Indiana State University, Terre Haute, IN; personal communication; June 1, 1999), who analyzed similar witness tree documents in the Oconee River Valley in central Georgia, we interpreted these posts as trees selected from the immediate vicinity of the corners and therefore indicative of the species that were present at the time of the surveys.

Researchers often directly plot data points representing witness tree locations to describe presettlement vegetation (e.g., Hong et al., 2000). Such point coverage, however, does not adequately describe forest cover, which is continuous. To create a continuous vegetative surface, grids or polygon maps must be extrapolated from the survey points (White and Mladenoff, 1994; He and Ventura, 1995; Brown, 1998a,b; Hong et al., 2000).

Foster, Thomas, Bryan Black, and Marc Abrams. In press. A witness tree analysis of the effects of Native Americans on the Pre-European settlement forests in east-central Alabama. *Human Ecology*.

Foster, H.T., II and Abrams, M.D. 2004. Physiographic analysis of the pre-European settlement forests in east-central Alabama. *Canadian Journal of Forest Research*. Research 32: 2060-2075.





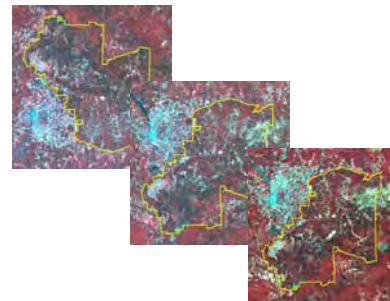
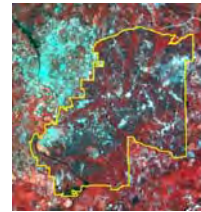
A methodology was developed to create the forest cover map by means of Arc/Info 7.21 TM, GRID TM, and ArcView 3.2TM. The survey points were first buffered with a radius of 160 m to ensure that each point represented both the corner marker and four witness trees located near most section corners (although a few such points represented as many as eleven trees). We recognize that this pixel size is quite coarse in relation to the satellite imagery. The resulting buffer polygons were then spatially joined to the survey points. Unique identifiers (poly-IDs) were assigned to each buffer polygon, and the point attribute table was edited to include the identifier of the polygon within which it was located. The descriptive information carried in the polygon attribute table was then reformatted to reflect tree type occurrence within the polygon (or survey point cluster). Tree species were assigned a value of either pine, deciduous, or other, with “other” representing small trees, shrubs, and ground cover. On the basis of the ratio of tree types within the cluster, individual polygons were categorized as pines, deciduous trees, mixed tree species (pine and deciduous), or other.

A raster grid was derived from the polygon coverage through the use of ARC INFO GRID. A grid cell size of 60 m was chosen to match the resolution of the other data in the study. A moving window capturing the neighborhood majority value was applied iteratively to the grid to create a continuous surface of vegetation types for Fort Benning. The final step was to overlay non-forest/cleared areas, which represent large Native American settlements as estimated from archaeological evidence (Foster et al., 2004). Because the locations of smaller settlements were not known, the amount of non-forested land is underestimated on the map.

The map of nineteenth century land cover produced from the witness tree data is useful even though the data do not provide continuous coverage and are highly variable. By augmenting the witness tree data with archeological evidence, the 1827 map could include areas of bare ground and sites developed for human use. In particular, the 1827 map shows that pine forest dominated the area even though native forests were actively burned as part of land clearing in the central region of Georgia. Similar witness tree data have been used to verify the presence of extensive pine stands in frequently burned areas of the Missouri Ozarks before European settlement (Batek et al., 1999) and in northern Florida (Schwartz, 1994).



- **LANDSAT 7 Enhanced Thematic Mapper (ETM+) July, 1999 (30m)**
  - 2 scenes (path/row)
    - 019/38
    - 019/37
- **North American Landscape Characterization (NALC) - Multispectral Scanner (MSS) triplicates (60m)**
  - 1974, 1983/86, 1991



To document more recent changes in land cover in the Fort Benning area, our study relied on remotely sensed data and created a series of four land-cover maps dating back to the 1970s. Two Landsat 7 Enhanced Thematic Mapper (ETM) images (i.e., path 019/row 037 and path 019/row 038) dated July 24, 1999, were acquired from the Environmental Characterization and Monitoring Initiative (ECMI) Data Repository (<http://sempdata.wes.army.mil>). The ETM images were used for making a current land-cover map of Fort Benning. The data had already been projected and mosaicked by means of nearest neighbor resampling.

North American Landscape Characterization (NALC) data, derived from Landsat Multispectral Scanner (MSS) imagery, were also used in this analysis (Lunetta and Sturdevant, 1993). Landsat MSS has a nominal resolution of 79 m, so we resampled the NALC data to a 60-m resolution. The NALC data set covering the Fort Benning area is composed of triplicates dated 1974, 1983/86, and 1991 for two scenes (i.e., path 019/row 037 and path 019/row 038). The two scenes for each time period had to be mosaicked in IMAGINE™ before the classification process could begin. The two scenes comprising the mosaic for the 1980s were made in different years; however, given the nature of the landscape and method of comparison used, this time interval was considered acceptable, and the date of mosaic will be referred to as “1983.”



## Reference Data

- February 1999 color digital orthophoto
- Land Cover Trend Analysis Data (LCTA)
- Streams



The four data sets (for 1974, 1983, 1991, and 1999) described in the previous paragraphs were then classified to create land-cover maps. Unsupervised classification, which creates a user-defined number of classes based upon spectral response, was used in-house to create 45 spectral classes from the imagery. We combined these 45 classes into six land-cover classes, using a 0.5-m resolution digital color orthophoto (1999) (Obtained from the web site for the wider project: <http://sempdata.wes.army/>) and Land Condition Trend Analysis (LCTA) point data (1991) (Diersing et al., 1992; Jones and Davo, 1997) as reference data. The LCTA protocol uses a modified point intercept method to quantify vegetation cover, ground cover, and disturbance at 1-m intervals on a 100-m line transect with transects randomly placed within a stratified random sampling scheme within an installation. Density and size distribution of woody vegetation are quantified in 600 m<sup>2</sup> plots that are aligned with each transect. The LCTA data tend to underestimate species richness (Prosser et al., 2003) yet provide consistent measures of changes in disturbance, canopy cover and bare ground (Anderson, 2002). In our analysis, the six land-cover classes are water, pine forest, mixed forest (deciduous and pine, areas of sparse forest cover, or areas of transition between forest and non-forest), deciduous forest, barren or developed land, and non-forest (areas cleared of forest vegetation but with some ground cover that may be grass or transitional areas). These classes are not only distinguishable in the maps; they also reflect ecological conditions of importance to resource managers. To remove any confusion between vegetation and water classes, we created and applied a water mask, using coincident pixels from the classified imagery and the coverage of lakes and major streams. A combination of ARC INFO 7.2.1<sup>TM</sup>, GRID<sup>TM</sup>, ArcView 3.2<sup>TM</sup>, and ERDAS IMAGINE 8.2<sup>TM</sup> software allowed us to derive land cover from the satellite imagery.



- **Arc/Info 7.2.1, Arcview 3.2, Erdas Imagine**
- **Mosaicked/projected imagery using nearest neighbor resampling**
- **Unsupervised classification - 45 classes**
- **Used reference data to combine 45 classes into 6 land cover classes**

Because all maps are highly generalized representations of reality and contain some error (e.g., Foody, 2002; Brown et al., 1999; Dicks and Lo, 1990; Maling, 1989; Smits et al., 1999), accuracy assessments were conducted to determine how accurately our classification portrayed land cover. We note that accuracy statements that accompany maps of large areas may be erroneous, vague, or nonsite-specific (Foody, 2002). Data, therefore, should be reviewed carefully in the context of its intended use (Fosnight and Greenlee, 2000). LCTA data provided some ground truth information to evaluate the forest type classes, but a preliminary comparison using the LCTA data proved to be variable due to inconsistencies in scale, definitions of cover types, and the sampling dates of the field data. An accuracy assessment was conducted for the 1999 land-cover classification using a 1999 0.5-m digital color orthophoto. The orthophoto was deemed an appropriate source of reference data in the absence of ground truth data. Fifty points per land-cover class were randomly generated and blindly classified on the basis of interpretation of the orthophoto and use of techniques described by Congalton and Green (1999). To facilitate comparison with the other land cover maps, we resampled the 1999 classification to 60-m resolution using nearest neighbor resampling.



- **Created and applied a water mask in GRID to eliminate confusion between water and vegetation classes**
- **Removed clouds and cloud shadows**
  - **digitized shapes**
  - **overlaid shapes with orthophoto**
  - **refined boundaries and determined appropriate land cover class**
  - **plotted shapes with classified image, adjusted boundaries and merged**

Clouds were not an issue with the MSS data since USGS picks the triplicate data dates upon getting the best, cloud-free data possible whereas the 1999 Landsat 7 Enhanced Thematic Mapper (ETM) images were selected by the Fort Benning resource managers. Therefore we focused on the 1999 image in considering ways to eliminate the presence of clouds and shadows, which were erroneously classified as barren/developed, the cloud and shadow areas were digitized and then overlaid with the 1999 orthophoto. New shapes that more accurately reflected the nature of the vegetation in these cloud-affected areas were digitized on-screen. These polygons were coded with a vegetation type according to interpretations drawn from the orthophoto. The resulting shapefile was plotted with the classified image and adjusted to maintain continuity and blend with neighboring pixels. A grid was created from the cloud-affected-area shapefile, and the classified land-cover image was also converted into a grid. Values from the cloud/shadow grid were used to mask the cloud-affected areas of the classified grid and create an improved land-cover map.



## Land Cover Classes

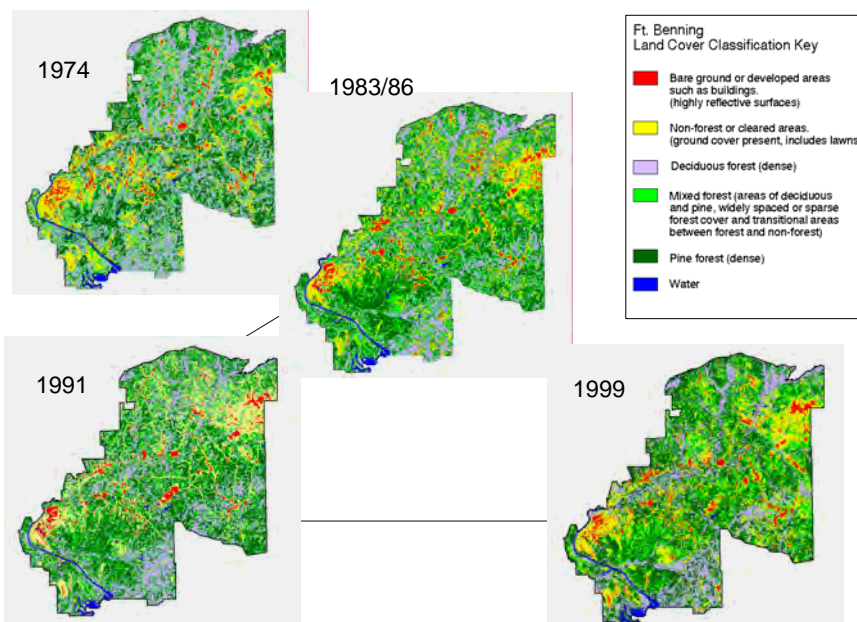
- Bare ground or developed areas such as buildings (highly reflective surfaces)
- Non-forest or cleared areas (ground cover present, includes lawns, and transitional areas)
- Deciduous forest (dense)
- Mixed forest (areas of deciduous and pine, widely spaced or sparse forest cover, and transitional areas between forest and non-forest)
- Pine forest (dense)
- Water



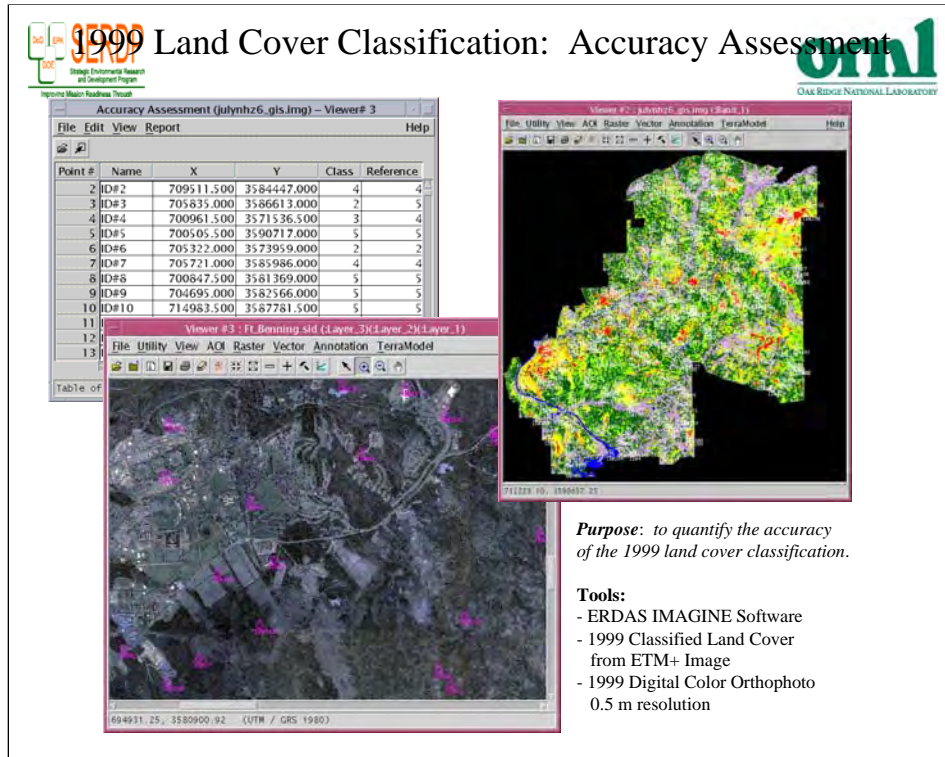
- **The ability of this classification to differentiate forest classes is a concern**
- **Expected errors:**
  - **Pine and deciduous forest misclassified as mixed forest (over-represent mixed forest, under-represent pine and deciduous forest)**

Results were calculated with the use of the default parameters of ATtILA and FRAGSTATS (e.g., 1 pixel minimum patch size, 0 pixel separation of patches, 10 pixel search radius, and 9 pixel moving window). Metrics were calculated for the historical (1827) data, the 60-m NALC data (1974, 1983/86, and 1991), and the ETM+ (1999) land-cover classification. As mentioned earlier, the ETM+ classification was resampled to a 60-m resolution for comparison with the NALC and historical data. Because the resolution of the source data differed originally, caution is necessary in interpreting the results of the analysis. In particular, the magnitude of change reflected by these metrics may be somewhat exaggerated because the historical data are coarse. On the other hand, the magnitude of the change may be underestimated as a result of the misclassification of pine pixels into the mixed forest class. Further caution must be exercised because of the uncertainty associated with the “mixed forest” class. Because they have a distinctive signature, bare areas are rarely misclassified.









Two ways of considering the accuracy of the maps produced from the imagery as compared to the orthophotographs were explored. The omission error quantifies how well cover types identified on the orthophotograph are correctly identified on the map derived from remotely sensing imagery. In contrast, the commission error reveals how well the map depicts cover types on the ground (or in this case, the orthophotographs). Both perspectives are needed to understand the validity of maps.



**Accuracy Assessment -1999 Landsat ETM Classification - resampled to 60m resolution**

|                                     | <i>Reference</i> | <i>Class</i> | <i>#Correct</i> | <i>Producer' s</i> | <i>User's</i> |
|-------------------------------------|------------------|--------------|-----------------|--------------------|---------------|
| Class 3<br>(dense deciduous forest) | 43               | 50           | 41              | 95.35%             | 82.0%         |



Example: Producers versus user's accuracy

Producers: 95.35% of deciduous areas (as identified on orthophoto) have been correctly identified as deciduous on map

Users: 82.0% of areas called deciduous on classified map are deciduous on ground (or on orthophoto in this case)

The ability of the classification to differentiate among forest classes varied within the time series. However, the mixed forest class is the most reliable classification for the 1827 period since the presence of some deciduous and coniferous trees can be overestimated from the survey data. Thus, differences in the area of the “mixed forest” class may be more indicative of changes in the collection (sensors) and methods of processing of the satellite data than of actual changes in forest composition. The 1991 image was classified first and then used as a “reference base” for the earlier images because appropriate reference data were not readily available for dates earlier than 1990. Similarly, no adequate data were available to perform a rigorous accuracy assessment on these earlier classifications. Therefore, some uncertainty is associated with the data. The goal of the project was to evaluate landscape indicators. The quality of the classification is open to improvement if more resources become available or if additional ancillary data is used in the analysis (Klema, 2001). Other papers examining the relationship between accuracy and indices focus on indicators of vegetation cover [e.g., NDVI (Liu et al., 2004)] or particular land-use classes as indicators of landscape condition (e.g., Berlanga-Robles and Ruiz-Luna, 2002).



|  <b>Accuracy Assessment -1999 Landsat ETM Classification - resampled to 60m resolution</b>                          |                  |              |                 |                   |               |
|---|------------------|--------------|-----------------|-------------------|---------------|
|   | <i>Reference</i> | <i>Class</i> | <i>#Correct</i> | <i>Producer's</i> | <i>User's</i> |
| Class 1<br>(bare ground/developed)  | 45               | 50           | 44              | 97.78%            | 88.0%         |
| Class 2<br>(non-forest vegetation)  | 51               | 50           | 40              | 78.43%            | 80.0%         |
| Class 3<br>(dense deciduous forest)   | 43               | 50           | 41              | 95.35%            | 82.0%         |
| Class 4<br>(mixed forest/sparse forest/transitional)  | 64               | 50           | 38              | 59.38%            | 76.0%         |
| Class 5<br>(dense pine forest)  | 44               | 50           | 36              | 81.82%            | 72.0%         |
| <p> <i>Kappa coefficient = 0.7458</i><br/> <i>Class 1 = 0.8537</i><br/> <i>Class 2 = 0.7487</i><br/> <i>Class 3 = 0.7826</i><br/> <i>Class 4 = 0.6774</i><br/> <i>Class 5 = 0.6602</i> </p> <p> <b>Overall accuracy = 75.20%</b><br/> <b>(Accuracy of 30 m resolution = 85.6%)</b> </p> |                  |              |                 |                   |               |

Overall accuracy for the 30-m classification was 85.6% and for the 60-m resolution classification was 75.20% (Table 1). The commission error was lower than the omission error for all land-cover types except mixed forest. This difference may be because the remote sensing product overestimated the area of mixed forest. The classification of forest classes was expected to contain errors of omission (errors of exclusion) in the pine and deciduous classes and of commission (errors of inclusion) in the mixed classes. Commission occurs when a class on the map includes pixels that should belong to another class (Congalton and Green, 1999). Specifically, the “mixed forest” class may contain pixels of deciduous forest and pine forest, resulting in an underestimation of these classes and an overestimation of mixed forest.

Determining the accuracy of the current map was a necessary step in assigning appropriate confidence in the map products. The process of accuracy assessment highlighted the mixed forest, which may be overestimated. Accuracy assessment is also a useful communication tool in conveying the information revealed by the landscape metrics to decision makers, for it quantifies the classification inaccuracies. It opens the door for mentioning that map accuracy can vary spatially and may be related to land-cover type, terrain, landscape complexity, and land-use patterns (Steele et al., 1998).

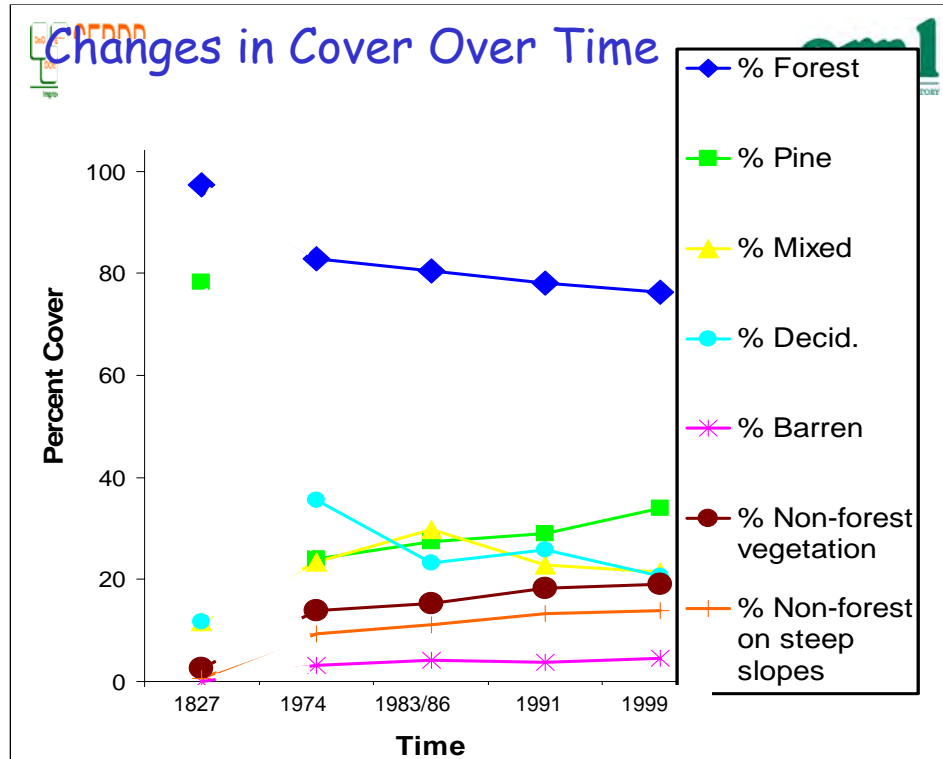


- **Indicators of fragmentation**
  - Describe the size, shape, and distribution of relatively homogenous “patches” of vegetation or land cover
- **Software**
  - **Analytical Tools Interface for Landscape Assessments (ATtILA)**
    - An ArcView extension developed cooperatively by the Environmental Protection Agency and the Tennessee Valley Authority
    - Can provide metrics by reporting unit
  - **FRAGSTATS**
    - developed by Kevin McGarigal and Barbara Marks
    - Can report metrics by cover type

Indicators of fragmentation were examined to consider ways they can improve understanding of land-cover changes at the landscape level. Landscape metrics were calculated through the use of two computer programs, FRAGSTAT 3.1 (McGarigal and Marks, 1995) and the Analytical Tools Interface for Landscape Assessments (ATtILA) (Ebert et al., 2001). The analysis concentrated on those indicators that differentiated between broad-scale patterns and revealed independent information (Tischendorf, 2001). ATtILA provides a relatively new suite of tools and is available as an ArcView<sup>TM</sup> extension. It requires Spatial Analyst<sup>TM</sup> and computes landscape metrics based on a land-use/land-cover grid, elevation and slope grids, a streams line theme, a roads line theme, a population polygon theme (county, track, or block), and a precipitation grid. The following ATtILA metrics were of particular interest to this project: land-cover proportions by reporting unit, the amount of non-forested land encountered on steep slopes, diversity metrics, and forest patch metrics. FRAGSTATS (McGarigal and Marks, 1995), however, was used to compute numerous other landscape metrics. Those reported here include total edge (m); number of patches; an index for the largest patch; mean, range, and coefficient of variation (CV) for the patch area; mean patch area ratio, and the CV of the Euclidean nearest neighbor distance, shape range, and clumpiness. These metrics were selected because they represented statistically significant changes over time in land cover for Fort Benning (Olsen et al., in review). The distance between patches was examined, for it can be important for species that move among a single patch type. The shape index corrects for the size problem of the perimeter-to-area ratio and thus is a measure of overall shape complexity. Clumpiness indicates the proportion of adjacent cells have identical land cover types. Output from each software package was compared and summarized. ATtILA and FRAGSTATS produced the same information describing patch dynamics and landscape composition; therefore, only ATtILA output is presented for the metrics.

The ATtILA and FRAGSTATS packages each offered some advantages. While there is some replication of metrics, FRAGSTATS calculates some metrics that are not standard output of ATtILA and can output information for specific cover types in addition to summary information for the entire landscape. A benefit of using the ATtILA program is that it allows the calculation and summation of metrics by reporting unit (e.g., catchments or training compartments). Although metrics for the entire installation are useful, metrics for subareas, such as the training compartment, are also effective descriptors, especially as certain military activities are designated to occur only in specific training compartments. Characterizing ecological impacts by compartments allows range managers to adjust training regimes to reduce future ecological effects.






Pine forests dominated the landscape at Fort Benning in 1827 . At that time over 75% of the land area was in pine forest with the second highest category, mixed forest, covering only about 12%. While these data are extremely generalized, the dominance of pine on the historical landscape is supported by the fact that over 90% of the soils at Fort Benning can support longleaf pine (Dale et al., 2002). Furthermore, archaeological evidence suggests that Native Americans almost never cleared forests except adjacent to their settlements (Silver, 1990; Foster et al., 2004).


The four-part time series of contemporary land-cover maps from 1974 to 1999 shows persistent landscape features as well as changes in land-cover composition over time. Changes in areas of deciduous forest are more commonly associated with riparian areas. Areas of bare ground have been fairly stable throughout the period of interest. In recent decades, Fort Benning has experienced a gradual decrease in forest populations along with an increase in non-forest vegetation.

The prevailing trend for recent decades is an increase in pine forest and a decline in deciduous forest area. The trend may result from ongoing forest management practices at Fort Benning that are aimed at establishing more areas of longleaf pine forest that can support the federally listed red-cockaded woodpecker. These practices include planting longleaf pine seeds and seedlings, as well as regularly setting controlled ground fires, which eliminate hardwoods and other pines while allowing the grass stage of longleaf pine saplings to survive.

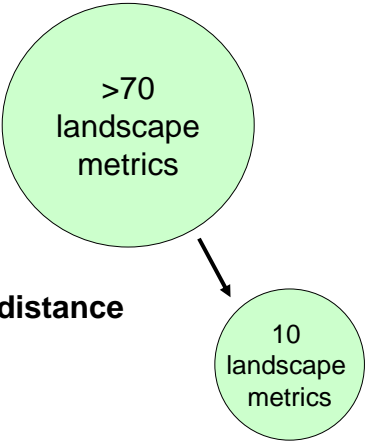




**The most useful metrics for distinguishing changes in land cover classes at Fort Benning:**



- Percent cover
- Total edge (with border)
- Number of patches
- Mean patch area
- Patch area range
- CV of patch area
- Perimeter area ratio
- Euclidean nearest neighbor distance
- Shape range
- Clumpiness



[Choice of metric depends on question]

Dale, V.H., Olsen, L.M. and H.T. Foster. In review. Landscape patterns as indicators of ecological change at Fort Benning, GA. Journal: *Land Use and Urban Planning*

[Continued description of previous slide]

The proportion of bare areas has remained relatively stable between 1974 and 1999. Mechanized military training at Fort Benning causes areas to become bare and remain barren of vegetation. Because tracked vehicles are especially detrimental to vegetation, training with tracked vehicles is restricted to certain sacrifice areas. The concept is that it is better for the overall environment to sacrifice a few areas while maintaining high quality ecological conditions in other areas. Hence at Fort Benning the locations of sacrifice areas have not moved in recent decades.

Dramatic changes to the Fort Benning landscape occurred between 1827 and 1974. By the early 1970s, pine forests had declined to about 25% cover, and deciduous forests dominated the landscape. Mixed forest slightly increased in cover. Areas with non-forest vegetation increased to about 15% of cover. Our analyses reveal how the landscape, once dominated by almost uninterrupted pine forest, transitioned into a mosaic of many different land uses and cover types such as those that occur elsewhere in the contemporary southeastern United State. (Frost, 1993). Even so, the Fort Benning installation still supports more pine forests (more than 35% of the area) than most places in the southeast (Frost, 1993). These changes in pine distribution largely explain the current distribution of red-cockaded woodpecker. When pine forest were abundant in the Fort Benning region, the red cockaded woodpecker, which nests in living pine trees, were abundant. However, the expansion of agriculture in the southeastern US in the mid to late 1800s and early 1900s reduced the distribution of pine forests (Kane and Keeton 1998), which greatly contributed to the demise of red cockaded woodpeckers. Yet, the situation at Fort Benning was different. These maps reveal the success of management programs at Fort Benning in expanding the distribution of pine forests in recent decades. The broad distribution of so many longleaf pine trees is a key reason that Fort Benning is a major component of the Fish and Wildlife Service's management plan for red cockaded woodpecker. Characteristics of the mixed forest are important to monitor, for they reveal information about areas that could be managed to support pine forests.

While the percentage of non-forest vegetated land has slightly increased, the metrics show that number of non-forest vegetated patches has increased tremendously in the past 25 years. Consequently, the size of these patches has decreased. Altered management has likely constrained certain land uses to smaller geographic areas.



| Metric                           | 1827           | 1974          | 1983          | 1991          | 1999          |
|----------------------------------|----------------|---------------|---------------|---------------|---------------|
| <b>Number of patches</b>         | <b>19</b>      | <b>11859</b>  | <b>16021</b>  | <b>16790</b>  | <b>17450</b>  |
| <b>Largest patch index</b>       | <b>73.517</b>  | <b>0.0581</b> | <b>0.0628</b> | <b>0.0351</b> | <b>0.027</b>  |
| <b>Mean patch area (ha)</b>      | <b>3038.31</b> | <b>1.08</b>   | <b>0.94</b>   | <b>0.95</b>   | <b>1.01</b>   |
| <b>Patch area range (ha)</b>     | <b>54280.8</b> | <b>30.6</b>   | <b>33.12</b>  | <b>18.36</b>  | <b>14.04</b>  |
| <b>Patch area CV</b>             | <b>397.75</b>  | <b>108.30</b> | <b>124.54</b> | <b>108.82</b> | <b>108.44</b> |
| <b>Mean perimeter/area ratio</b> | <b>290.86</b>  | <b>525.34</b> | <b>554.29</b> | <b>547.01</b> | <b>541.53</b> |
| <b>CV of Euclidean NND</b>       | <b>70.83</b>   | <b>34.03</b>  | <b>29.34</b>  | <b>28.61</b>  | <b>24.47</b>  |
| <b>Shape range</b>               | <b>2.86</b>    | <b>2.83</b>   | <b>3.10</b>   | <b>2.17</b>   | <b>2.21</b>   |
| <b>Clumpiness</b>                | <b>0.94</b>    | <b>0.13</b>   | <b>0.04</b>   | <b>0.02</b>   | <b>-0.02</b>  |

Subtle changes to land cover at Fort Benning occurred in recent decades. The area of pine forest has been gradually increasing in the past 25 years. Improved monitoring techniques coupled with an aggressive management strategy for perpetuating pine forest at Fort Benning (Waring et al., 1990) have likely been the cause for the increase in pine and the corresponding decrease in deciduous forest. This management strategy includes harvesting timber from stands other than longleaf pine and thinning and burning areas of longleaf pine (Haywood et al., 1998; Gilliam and Platt, 1999; Provencher et al. 2001).

Total edge was computed from slightly different data than the other metrics discussed here. While the others were computed from land-cover data that had been clipped to the boundary of Fort Benning, the edge metrics were calculated from land-cover data that extended beyond the boundary of the installation. This second approach allowed us to calculate the true edge of the class patches instead of accepting the edge as summarily defined by the legal boundaries of the installation.

Numerous landscape metrics were computed for each land-cover class through the use of FRAGSTATS (Table 2). We considered the metrics by each land cover type since each type reveals different information about resource management at Fort Benning. After a big increase from 1827 to 1974, the total amount of edge associated with bare/developed patches remained fairly stable throughout the recent decades. This result parallels other patch metrics for the bare/developed class, suggesting that overall impacts of training on bare ground have not changed much in recent years. However, the patch area coefficients of variation have increased, revealing more variability in size of the patches with time. Yet, the amount of edge associated with non-forest vegetation has been increasing probably because more such patches existed in 1999 as compared to 1974. The forest classes also experienced dramatic changes. The amount of edge associated with the pine forest class increased over the entire time period. The amount of edge associated with mixed forest class increased rather sharply between 1827 and 1974 but has been decreasing since 1983. The edge associated with deciduous forest has vacillated considerably: it increased sharply between 1827 and 1974, decreased abruptly between 1974 and 1983, then increased between 1983 and 1991, and decreased between 1991 and 1999. These abrupt changes in direction of the mixed and deciduous forest types may reflect management actions to promote longleaf pine; such actions have the effect of adding land to the pine forest class and removing land from the other two forest classes, thus changing their distribution on the landscape.

The number of patches associated with each land-cover class increased dramatically between 1827 and 1974 but thereafter fluctuated for bare areas and deciduous and mixed forest and continued to increase for pine and nonforested areas (Table 2). Decreases in edge associated with the bare areas and deciduous and mixed forest classes parallel decreases in the number of patches associated with the forest classes between 1991 and 1999. The larger change over the 137-year period from 1827 to 1974 partially reflects the different mapping technologies but also conveys how significantly human activities have affected the landscape. There was a recent upsurge in the number of bare and developed patches.

The largest patch index (LPI) represents the percentage of the landscape that contains the largest patch of each class and thus the dominance of a single large patch. The low values of this index (except for the high 73 value for pine forest in 1827) reflect the lack of dominance of any land-cover type (except when pine forests were common before European settlement). These large patches are important for organisms that require large habitats.



| Metric  | Year   |        |        |        |        |
|---|--------|--------|--------|--------|--------|
|   | 1827   | 1974   | 1983   | 1991   | 1999   |
| Total edge with border (km)                     | 68     | 610    | 847    | 720    | 989    |
| Number of patches                               | 9      | 2274   | 2905   | 2501   | 3460   |
| Largest patch index                             | 0.679  | 0.0149 | 0.0142 | 0.0149 | 0.0176 |
| Mean patch area (ha)                            | 173.32 | 0.95   | 0.89   | 0.98   | 0.85   |
| Patch area range (ha)                           | 480.24 | 7.56   | 7.2    | 7.56   | 9      |
| Patch area coefficient of variation (CV)        | 94.39  | 86.39  | 94.57  | 104.98 | 108.71 |
| Mean perimeter/area ratio                       | 73.60  | 528.63 | 547.66 | 540.89 | 567.74 |
| CV of Euclidean nearest neighbor distance (NND) | 255.22 | 68.22  | 61.59  | 62.66  | 60.94  |
| Shape range                                     | 1.20   | 1.45   | 1.35   | 1.45   | 1.65   |
| Clumpiness                                      | 0.93   | 0.28   | 0.26   | 0.28   | 0.25   |

Mean patch area and patch area range closely follow the trends of the LPI. Both metrics were highest for pine forests in 1827. All patches were larger in 1827 than in the late twentieth century. The many small patches in all categories in recent decades also result in a smaller patch range. Mean patch area was not highly variable through recent times or among classes because of the presence of many single pixel patches. This stable trend, together with a decrease in the number of patches that are not in pine forest or bare/developed lands and an increase in pine forest composition, indicated that areas of pine forest were getting larger and more continuous. The coefficient of variation of the mean patch size for the bare/developed land has been gradually increasing since 1974. The other land-cover classes vary in this metric for recent times.

Edge metrics are more meaningful, however, when interpreted in conjunction with other metrics, such as mean patch area. For example, while the number of pine forest patches and total edge has been increasing since 1991, mean patch area for pine patches has remained stable during the same period. Total edge can be important for species that occupy ecotones, the margins of ecological systems (e.g., forest fringes). Patch area is critical for species that require a minimum home range size.



## Landscape metrics for deciduous forest land cover

| Metric                             | 1827    | 1974   | 1983   | 1991   | 1999   |
|------------------------------------|---------|--------|--------|--------|--------|
| <b>Total edge with border (km)</b> | 217     | 6600   | 3914   | 4528   | 4320   |
| <b>Number of patches</b>           | 47      | 18223  | 10670  | 12601  | 11257  |
| <b>Largest patch index</b>         | 1.376   | 0.0284 | 0.0209 | 0.0257 | 0.029  |
| <b>Mean patch area (ha)</b>        | 118.24  | 0.97   | 0.98   | 0.97   | 0.84   |
| <b>Patch area range (ha)</b>       | 1015.92 | 14.76  | 10.8   | 13.32  | 15.12  |
| <b>Patch area CV</b>               | 153.30  | 94.61  | 89.16  | 94.52  | 109.74 |
| <b>Mean perimeter/area ratio</b>   | 126.38  | 540.00 | 527.84 | 533.47 | 562.48 |
| <b>CV of Euclidean NND</b>         | 105.53  | 22.35  | 34.54  | 32.86  | 36.24  |
| <b>Shape range</b>                 | 1.50    | 2.01   | 2.06   | 1.88   | 2.36   |
| <b>Clumpiness</b>                  | 0.93    | -0.06  | 0.15   | 0.11   | 0.13   |

The mean perimeter-to-area ratio of the patches did not vary much between classes after 1974. The values were similar for pine, deciduous forest, bare/developed, and non-forest vegetation, while values for mixed forest were consistently higher than the rest. These similarities can be explained by the influence of single-pixel patches (small patches) upon this metric. Mixed forest is associated with the highest values because, as a transitional class, it is most often represented as single pixel or small patch. Deciduous forest is mainly found in riparian areas in large, contiguous patches. Bare/developed and non-forest patches are often the result of some type of managed land use and tend to occur in relatively simple and confined shapes. Overall, there was not much variability within patch level metrics.

The coefficient of variation (CV) of the Euclidean nearest neighbor distance accounts for variability in measures of the edge-to-edge distance (m) between patches of the same class. There was little within-class variation between 1974 and 1999. Bare/developed areas were the most widely spread patch type on the landscape followed by non-forest patches. These classes were largely associated with managed land use and comprised a smaller percentage of total landscape composition and number of patches. The forest class values were lower, indicating less distance between forest patches, which were also more prevalent on the landscape at Fort Benning.

Patch shape index-range increased between the 1827 map and recent maps for all classes except pine forest; this increase is a measure of the greater complexity in the shape of areas in land-cover types other than pine forest. After 1974, the shape range index was consistent for the bare/developed areas, but it had a peak in 1983 for the deciduous, mixed, and pine forests. At that same time the non-forest vegetated areas had a dip in their shape range index. These inflection points suggest that the land pattern was most complex in 1983.

The clumpiness index is a measure of patch aggregation. A value approaching 1 indicates a greater degree of patch clumping; zero signifies a random distribution of patches, and -1 indicates disaggregated patches. All land-cover types were more aggregated in 1987 than in other maps for recent decades. The data included negative values for mixed forest from 1974 to 1991, for deciduous forests in 1974, and for pine forests in 1999. Thus the degree of aggregation has fluctuated over time.



## Landscape Metrics for all of Fort Benning

| Year | Total edge with border (km) | Number of patches | Patch density | Largest patch index |
|------|-----------------------------|-------------------|---------------|---------------------|
| 1827 | 6624                        | 131               | 0.1774        | 73.5166             |
| 1974 | 49405                       | 61312             | 115.0106      | 0.0581              |
| 1983 | 48498                       | 61256             | 114.9055      | 0.0628              |
| 1991 | 52672                       | 62254             | 116.7626      | 0.0351              |
| 1999 | 93360                       | 61837             | 115.9954      | 0.0304              |

Landscape metrics can be useful indicators of overall landscape fragmentation and diversity. Most landscape level metrics changed considerably between 1827 and recent decades. Total edge increased more than seven fold between 1827 and 1974. The number of patches and patch density also increased over this 147-year period, and the largest patch index inclined dramatically. In other words a few, large patches occurred in the landscape in 1827, as is evident in a visual examination of the map.

Less change occurred from 1974 to 1999. As with the shape range index, there was an inflection point in 1983 for the landscape metrics for the entire installation. Compared with the other maps for recent decades, the 1983 map had fewer and larger patches with less edge. The amount of edge increased again after 1983 as the largest patch index declined. However, the trends in edge are not consistent with changes in the total number of landscape patches or patch density during the same time period. In general, one might expect the number of patches to increase with total edge, however, the trends do not necessarily agree. Clearly, this indicates a complex landscape.

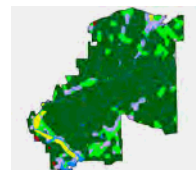
Furthermore, the patch-level metrics show that increases in the number of patches associated with certain land-cover classes are tempered by decreases in the number of patches associated with other land-cover classes throughout this time period (Table 2), resulting in a fairly stable number of patches at the landscape level. This change demonstrates the necessity of examining the patch-level metrics to get a complete picture of landscape change.



# Submitted Historical Data, Metadata and Reports to the SEMP- ECMI Data Repository



Witness Tree - Point Data



Historical Land Cover

Data from the all aspects of the indicator study have been placed in the SEMP data repository.



## Major Results

- Vegetation maps developed from different data sources can be compared
- Changes over time occurred in the vegetation at Forest Benning
  - **1827 to 1974**
    - **Percent cover of forest declined**
    - **Pine coverage decreased**
    - **Increase in deciduous, mixed and nonforest cover**
  - **1974 to 1999**
    - **Gradual increase in the nonforest vegetation (esp. on slopes)**
    - **Decline in cover of forest vegetation**
      - Increase in pine cover
      - Decrease in deciduous cover

(<http://www.esri.com/library/userconf/proc01/professional/papers/pap765/p765.htm>)

The selection of methods and landscape metrics for analysis of land cover changes was a key focus of this analysis. Landscape metrics are essential to quantifying changes in land cover. Of the large set of metrics available in the FRAGSTATS and ATtILA software, we found the most useful metrics for depicting changes in land cover at Fort Benning to be percent cover, total edge with border, number of patches, mean patch area, patch area range, the coefficient of variation in patch area, the perimeter-to-area ratio, Euclidean nearest neighbor distance, shape range, and clumpiness. While these last two metrics show little change over time, they distinguished between land-cover categories quite well. In terms of the entire landscape of Fort Benning, the metric for total edge changed most over recent time periods and was useful for analysis.

Data collected for disparate purposes can be used to help develop an understanding of land-cover changes over time and are often necessary to further our knowledge of historic conditions on a given landscape. For the entire Fort Benning landscape, the values of metrics for 1827 were very different from the values for recent decades. While the changes between 1827 and 1974 may be somewhat exaggerated due to data constraints, we can conclude that the nineteenth century landscape at Fort Benning was composed largely of uninterrupted pine forest with some deciduous forests found in riparian corridors and some open areas associated with Native American settlements. Land cover and land use in the 1970s were considerably different. Following decades of farming, military training activities had a pronounced affect upon the landscape. Heavy training activities resulted in areas of sparse land cover and bare ground. Interestingly, these areas have largely persisted on the landscape throughout the 1980s and 1990s. This result not only emphasizes the lasting footprint that military activities have on the landscape but also highlights the efforts made by management to confine heavy training exercises to certain sacrifice areas. Another interesting trend occurred in the 1990s. Pine forests have been on the rise as is reflected in both landscape composition and patch dynamics such as largest patch size, number of patches, and total edge. Management efforts at Fort Benning have focused upon managing for longleaf pine. These efforts appear to be decreasing hardwood invasion in favor of pine species in many areas on the installation.

Examining a suite of landscape metrics over time was useful for summarizing, describing, and assessing land-cover change at Fort Benning. The FRAGSTATS and ATtILA programs were relatively simple to use and provided information pertinent to understanding and managing the land. Therefore, we encourage resource managers to use landscape metrics to analyze changes in patterns of land cover over time to examine how human activities have affected an area.

At Fort Benning, these results can be used to understand the broad-scale effects of existing training and forest management practices and to help design new military training areas.



# Technology Transfer of Ecological Indicators

## Catchment-Scale Indicators Evaluation of stream measurements

Patrick J. Mulholland, Principal Investigator  
Oak Ridge National Laboratory  
Jack W. Feminella, Principal Investigator  
Auburn University





## Catchment-Scale Indicators

### Evaluation of stream measurements

#### **Principal Investigators:**

Patrick J. Mulholland (Oak Ridge National Laboratory)

Jack W. Feminella (Auburn University)

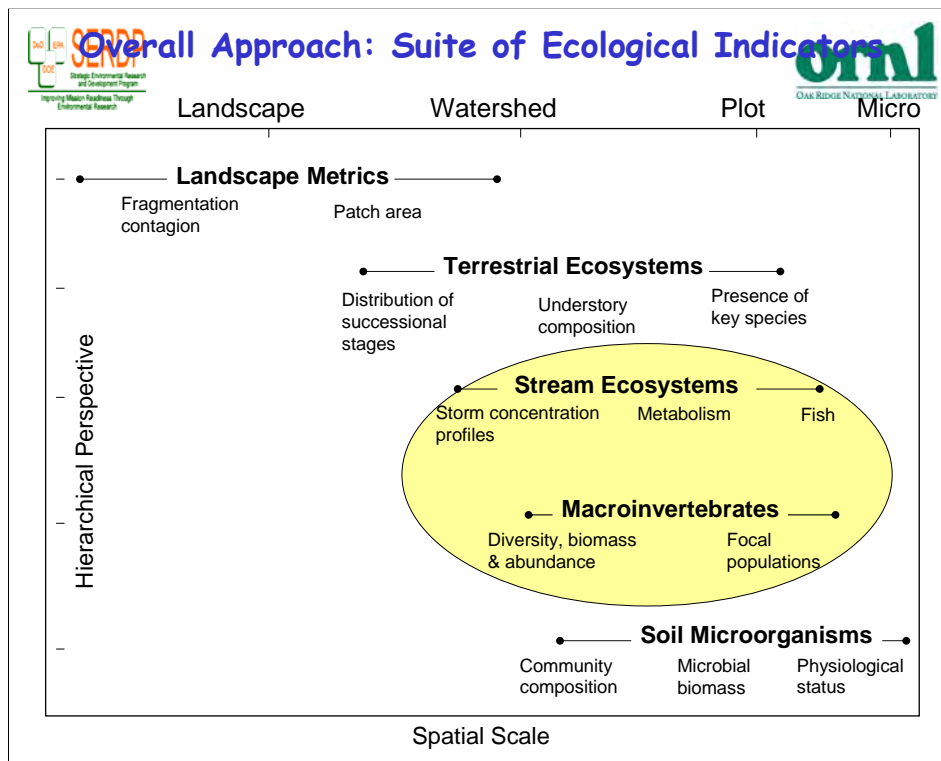
#### **Post-doctoral fellow:**

Jeffrey Houser (Oak Ridge National Laboratory)

#### **Students:**

Kelly Maloney (Auburn University)





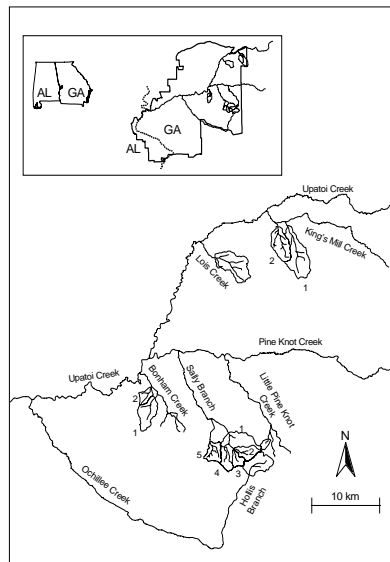
The response of stream ecosystem and macroinvertebrate indicators are part of a larger project designed to investigate indicators that would be useful to augment current sampling regimes at military bases and typical of other actively managed sites.



## Catchment-Scale Indicators

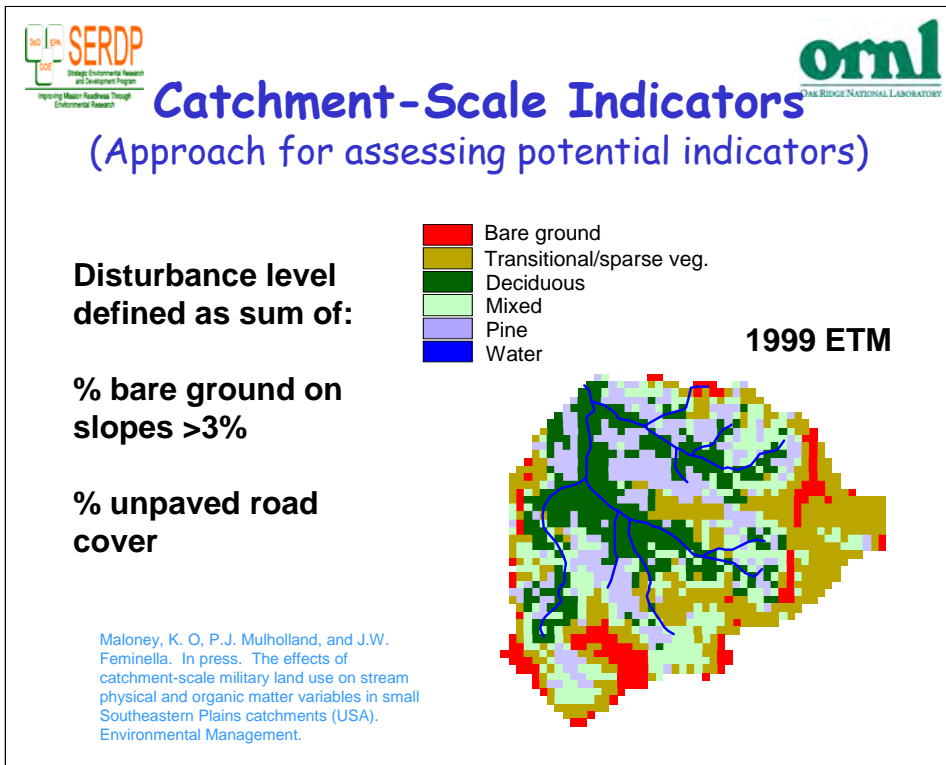
### Site locations

- 12 catchments on the eastern half of Fort Benning
- Streams were small, sandy bottomed, 1<sup>st</sup>-3<sup>rd</sup> order streams
- Catchment size ranged from 0.33 to 5.43 km<sup>2</sup>



Sites used for the evaluation of stream-based catchment scale indicators.



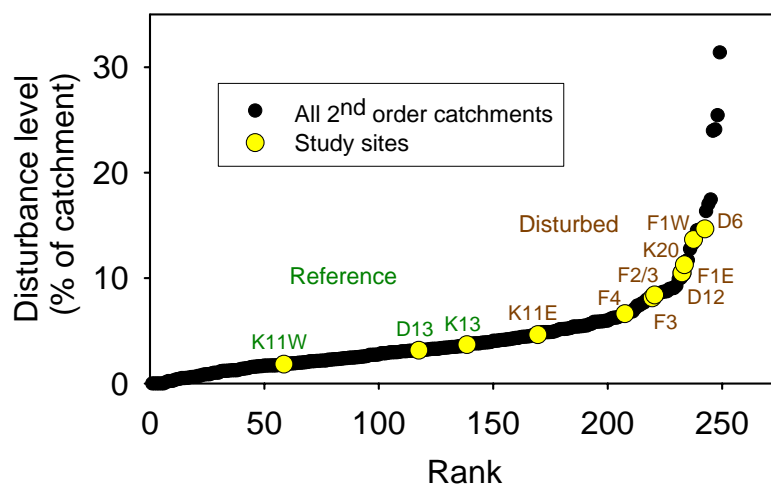


Because headwater streams are tightly linked to associated catchments they effectively integrate effects at catchment scales. Therefore, we used the relationships between potential stream indicators and catchment-scale disturbance levels (related to vegetation and soil disruption) as a basis for assessing indicators of disturbance. We quantified land use within study catchments using geographic information system (GIS) datasets (i.e., streams: 1:24,000, 1993 coverage; soils: 1:20,000, 1998; roads: 10-m resolution, 1995), digital orthophotographs (1:5,000, July 1999), digital elevation models (DEMs, 1:24,000, 10-m grid size, 1993) and Landsat imagery (28.5 m, July and December 1999). We processed data sets with catchment boundaries using ArcView© software (Environmental Systems Research Institute, Inc., Redlands, California). Landsat imagery, digital orthophotography, and DEMs were used to quantify the proportion of each catchment occurring in a particular land-use class on slopes >3%, using the ArcView extension Analytical Tools Interface for Landscape Assessments (ATtILA, Ebert and Wade 2000). We used 3% slopes as our threshold value because calculation of the universal soil loss equation indicated that slopes at or above this level showed the highest potential for increased annual soil loss in our study area (see GASWCC 2000).

Land-use and land cover categories used in our analyses were the proportion of bare ground and unpaved road cover (%BGRD) and the proportion of nonforested land in a catchment (%NF). The proportion of catchment on soils with >3% slopes and containing no vegetative cover was included in %BGRD, which also included unpaved roads. We quantified road cover by multiplying road length by average width; the latter was estimated in the field for the 2 different classes of unpaved roads found in our catchments: class-6 roads (6-m wide), and class-5 roads (20-m wide). The proportion of catchment on soils with >3% slopes that were vegetated but without dense forests, including grasslands, sparse vegetation, shrublands, and fields was incorporated into %NF.



## Catchment-Scale Indicators (Approach for assessing potential indicators)



We quantified our disturbance index for all 249 second order catchments at Fort Benning. We sampled 12 first to third order catchments that spanned the range of disturbance at Fort Benning. All 12 catchments were disturbed at some level, however, we generically classified the 3 least disturbed as reference and the remainder as disturbed.



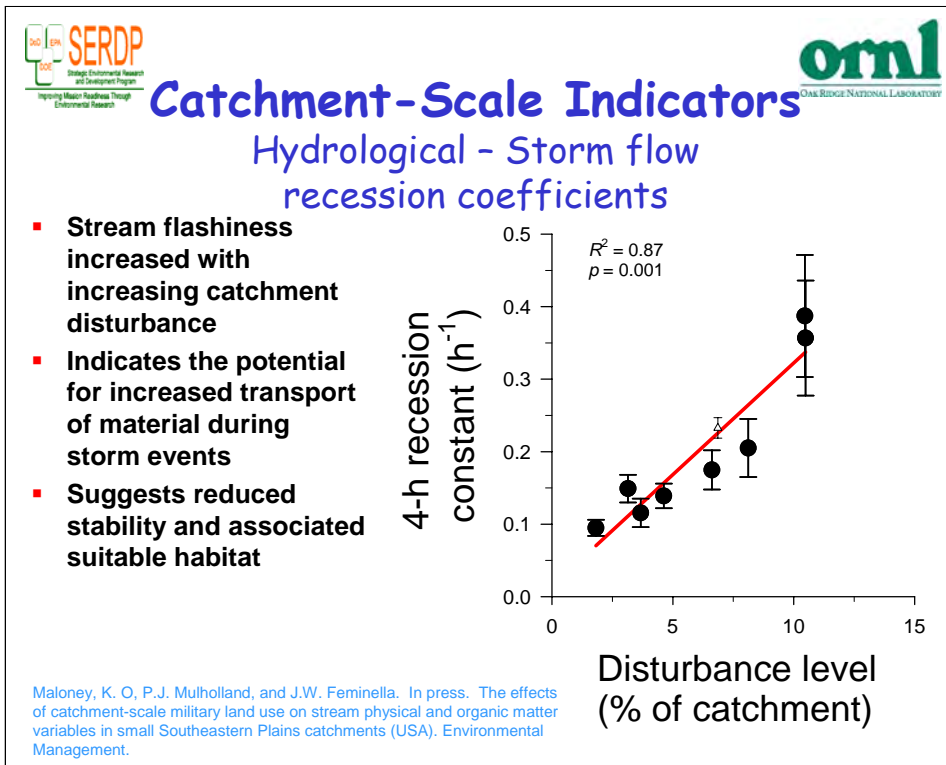
## Catchment-Scale Indicators

(Summary of applicable stream measurements)

- **Hydrological**
  - Storm flow recession coefficients
- **Chemical**
  - Suspended sediment concentrations (baseflow, storm)
  - Baseflow  $\text{PO}_4$  and DOC concentrations
  - Storm increases in  $\text{NO}_3$  and  $\text{PO}_4$  concentrations
  - Diurnal changes in dissolved oxygen concentrations
- **Biological Habitat**
  - Streambed stability
  - Coarse woody debris
  - Benthic particulate organic matter
  - Sediment particle size
- **Biota**
  - Macroinvertebrate assemblage
  - Fish assemblage

We took a multidisciplinary approach to identifying potential catchment-scale indicators of disturbance focusing on a multitude of stream variables. We tested the efficacy of hydrology, water chemistry (both non-storm and storm flow), “Habitat” (bed morphology, woody debris, organic matter), and biota (macroinvertebrates and fish) to indicate disturbance over a 3 year period from 2000 – 2002.





Storm hydrograph recessions integrate numerous sources of inflow (e.g., overland flow, interflow), and have been used by others to indicate stream flashiness (Rose and Peters 2001). Therefore, we quantified the rate of descent of the falling limb of several storm hydrographs in each stream as a relative measure of hydrologic flashiness. We estimated discharge from measurements of channel width, and water velocity and depth measured by an ISCO ultrasonic flow module (model 750) and series portable sampler (model 6700); depth and velocity were recorded every 30 min to 1 h. We computed recession constants for the initial portion of the hydrograph recession curves for each storm hydrograph as the slope of the natural logarithm of discharge over time during the first 4 hours following peak discharge (see Rose and Peters 2001). If the recession limb showed an obvious break in slope in <4 h, then we used data only prior to the break point to calculate recession constants. For 3 study catchments (LPK, SB1, SB5), we collected data for <3 storms, so we excluded these sites from analyses.

We observed a positive relationship between 4-h storm flow recession coefficients and catchment disturbance level indicating disturbance effects on catchment hydrology.



## Catchment-Scale Indicators

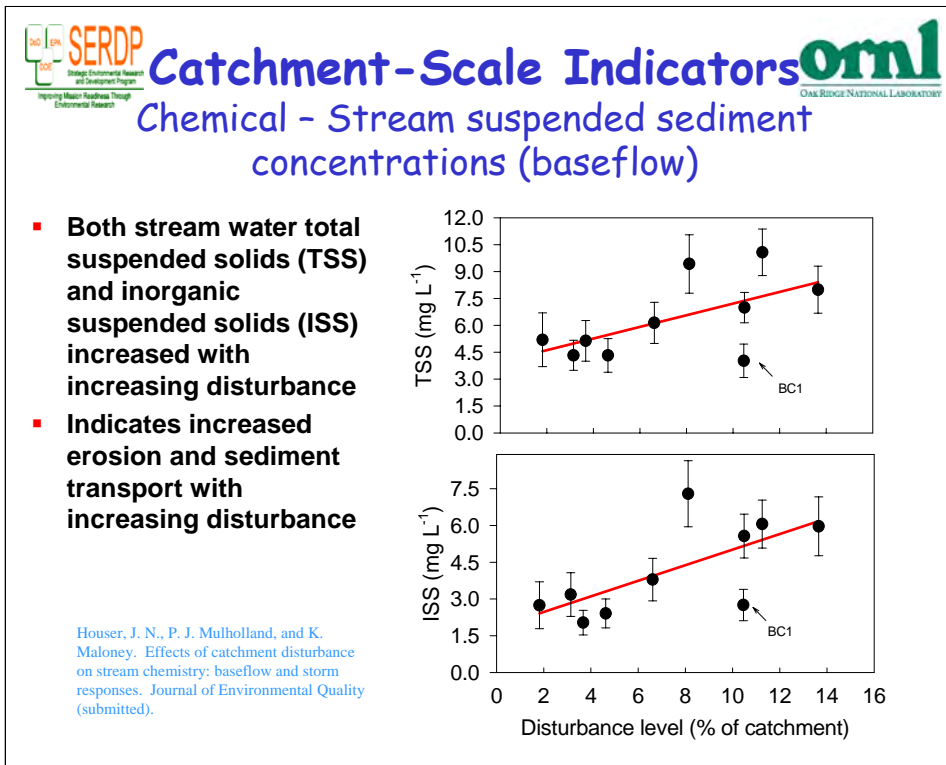
### Hydrological - Storm flow recession coefficients

#### Management implications

→ Coefficients consistently  $> 0.2 \text{ hr}^{-1}$  (1<sup>st</sup> and 2<sup>nd</sup> order streams) are found only in highly disturbed catchments ( $>10\%$  as bare ground and unpaved roads)

→ High coefficients indicate disruption of catchment hydrology producing “flashy” storm hydrographs which tend to produce more sediment transport and stream channel instability





Concentrations (mg/L) of suspended sediments (inorganic and total) in stream water are good measures of erosion and sediment transport from a catchment and the streambed instability that accompanies deposition of eroded sediments in streams. The observed positive relationship between total and inorganic suspended sediment concentrations (TSS and ISS, respectively) at baseflow and catchment disturbance level indicates increased erosion and sediment transport and increasing streambed instability with increasing disturbance.

#### Measurement and calculations:

(1) Filter 0.5 to 1.0 L of streamwater through pre-combusted and tared Whatman GFF filter, dry filter with sediments at 80°C for 2 d, reweigh (dry mass), combust filter with sediments at 500°C for 12 h, rewet combusted sediments on filter to replace water of hydration, dry at 80°C for 2 d and reweigh (ash mass)

(2)  $TSS = (\text{dry mass} - \text{tare mass}) / \text{volume filtered}$

$ISS = (\text{dry mass} - \text{ash mass}) / \text{volume filtered}$



## Catchment-Scale Indicators

### Chemical - Stream suspended sediment concentrations (baseflow)

#### Management implications

→ Stream TSS > 6 mg/L and ISS concentrations > 4 mg/L were only consistently observed in highly disturbed catchments

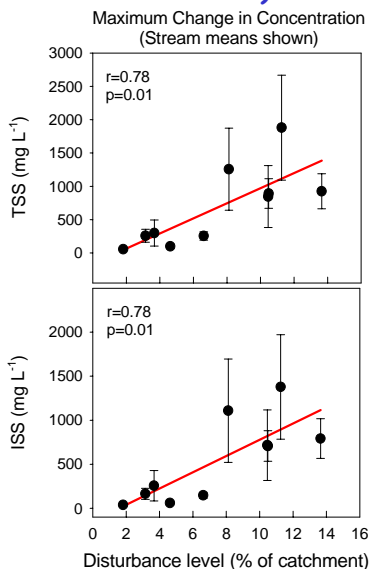
→ Disturbance levels > 8% of catchment as bare ground and unpaved roads appeared to be a disturbance threshold, above which stream TSS and ISS concentrations at baseflow are considerably higher.

→ Increased erosion and sediment transport from disturbance is evident even during baseflow, indicating disturbance produces highly unstable stream channels which will have significant negative effects on biota and biotic habitat.



## Chemical - Stream suspended sediment concentrations (storm increases)

- Both storm stream water total suspended solids (TSS) and inorganic suspended solids (ISS) increased with increasing disturbance
- Indicates increased erosion and sediment transport with increasing disturbance
- Variation in storm TSS and ISS increases was considerably greater in more highly disturbed catchments, indicating greater sediment instability is accompanied by reduced predictability



Increased concentrations (mg/L) of total suspended sediments (TSS) and inorganic suspended sediments (ISS) in stream water during storms are measures of active erosion and sediment transport. Observed positive relationships between storm increases in TSS and ISS concentrations and catchment disturbance level indicate increased erosion and sediment transport with increasing disturbance. Also, the variation in storm TSS and ISS increases was considerably greater in more highly disturbed catchments, indicating greater sediment instability is accompanied by reduced predictability.

### Measurement and calculations:

(1) Filter 0.1 to 0.5 L of streamwater through pre-combusted and tared Whatman GFF filter, dry filter with sediments at 80°C for 2 d, reweigh (dry mass), combust filter with sediments at 500°C for 12 h, rewet combusted sediments on filter to replace water of hydration, dry at 80°C for 2 d and reweigh (ash mass)

(2)  $TSS = (\text{dry mass} - \text{tare mass}) / \text{volume filtered}$

$ISS = (\text{dry mass} - \text{ash mass}) / \text{volume filtered}$



## Catchment-Scale Indicators

Chemical - Stream suspended sediment concentrations (storm increases)

### Management implications

→ Increases in storm TSS and ISS concentrations > 400 mg/L were observed consistently only in highly disturbed catchments. Variability in storm increases was considerably greater in highly disturbed catchments.

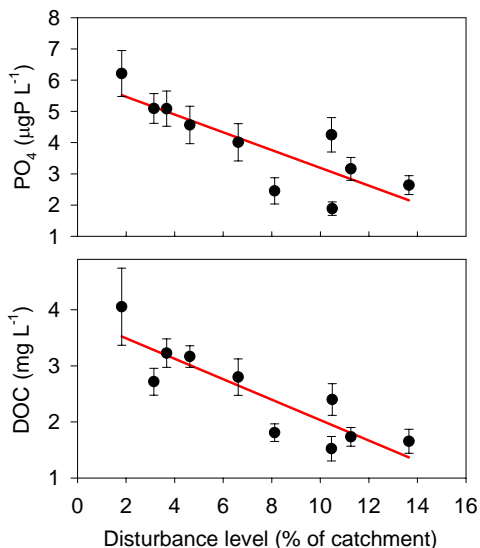
→ Disturbance levels > 8% of catchment as bare ground and unpaved roads appeared to be a disturbance threshold for storm increases in TSS and ISS concentrations (consistent with baseflow patterns).

→ Increased erosion and sediment transport from disturbance is particularly evident during storm periods and these episodes have lasting negative effects on biota and biotic habitat.



## Chemical - Stream $\text{PO}_4$ and DOC concentrations during baseflow

- Both baseflow phosphate ( $\text{PO}_4$ ,  $\mu\text{gP/L}$ ) and dissolved organic carbon (DOC,  $\text{mgC/L}$ ) concentrations in stream water decreased with increasing catchment disturbance
- Results may indicate increased sorption to clay-rich stream sediments or reduced leaching from upland soils with reduced microbial activity



Phosphate ( $\text{PO}_4$ ,  $\mu\text{gP/L}$ ) and dissolved organic carbon (DOC,  $\text{mgC/L}$ ) concentrations in stream water are indicators of biogeochemical processes in catchments and/or streams. We observed negative relationships between  $\text{PO}_4$  and DOC concentrations and disturbance level. We believe these are the result of increased sorption to clay-rich stream sediments and/or reduced leaching from upland soils due to lower organic matter content and reduced microbial activity. Because  $\text{PO}_4$  and DOC are important nutrients for stream biota, disturbance has resulted in an impoverishment of stream ecosystems.

### Sample processing and analysis:

- Collect water sample and filter at least 100 mL through pre-combusted Whatman GFF filter and store at least 50 mL of filtrate in acid-washed polypropylene bottles ( $\text{PO}_4$ ) or combusted ( $500^\circ\text{C}$ ) glass vials after addition of 2 drops of 6N HCl (DOC). Freeze  $\text{PO}_4$  or refrigerate DOC sample until analysis.
- $\text{PO}_4$  analysis: Molybdenum blue method using spectrophotometer with 10-cm pathlength cell (Standard Methods for the Examination of Water and Wastewater, 1992, 18<sup>th</sup> Edition, American Public Health Association)

DOC analysis: High temperature combustion method using total carbon analyzer (e.g., Shimadzu model 5000)



## Catchment-Scale Indicators

### Chemical - Stream $\text{PO}_4$ and DOC concentrations during baseflow

#### Management implications

→ Lower  $\text{PO}_4$  and DOC concentrations with increasing levels of catchment disturbance appear to be due to either: (1) increased erosion and deposition in stream channels of clay-rich sediments which have a high sorption capacity, or (2) a decline in leaching of these solutes from upland soils that have lower organic matter content and biological activity due to disturbance.

→ Lower stream  $\text{PO}_4$  and DOC concentrations will tend to reduce algal and microbial productivity due to intensified nutrient limitation. These effects would likely lead to reduced productivity of macroinvertebrates and fish.

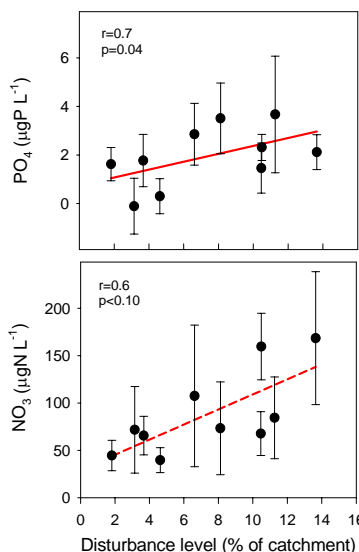
→ There was no evidence of a threshold disturbance level in the relationships between  $\text{PO}_4$  and DOC concentrations and disturbance level.



## Catchment-Scale Indicators

### Chemical - Increase in stream $\text{NO}_3$ and $\text{PO}_4$ concentrations during storms

- Positive relationships between the increase in  $\text{NO}_3$  and  $\text{PO}_4$  concentrations during storms and catchment disturbance level
- Results may indicate reduced retention of atmospheric inputs of  $\text{NO}_3$  in upland areas and mobilization of  $\text{PO}_4$  from sediments in ephemeral and perennial stream channels



Increases in streamwater  $\text{NO}_3$  ( $\mu\text{gN/L}$ ) and  $\text{PO}_4$  ( $\mu\text{gP/L}$ ) concentrations during storm periods are indicators of biogeochemical processes in catchments and/or streams. We observed positive relationships between the increase in  $\text{NO}_3$  and  $\text{PO}_4$  concentrations during storms and catchment disturbance level. We believe these are the result of reduced retention of atmospheric inputs of  $\text{NO}_3$  in upland areas and mobilization of previously sorbed  $\text{PO}_4$  from sediments in ephemeral and perennial stream channels.

#### Sample processing and analysis:

(1) Collect water samples during storm periods using an auto-sampler (e.g., ISCO) or manually at least hourly during rising limb, peak and initial falling limb of hydrograph, less frequently thereafter.

(2) Filter at least 100 mL of streamwater through pre-combusted Whatman GFF filter and store filtrate in acid-washed polypropylene bottles in freezer until analysis.

(3)  $\text{NO}_3$  analysis: Cu-Cd reduction followed by azo dye colorimetry using autoanalyser (e.g. Bran Luebbe AA3).

$\text{PO}_4$  analysis: Molybdenum blue method using spectrophotometer with 10-cm pathlength cell

(4) Calculate storm increase in concentration by subtracting concentrations measured on samples taken prior to storm.



## Catchment-Scale Indicators

### Chemical - Increase in stream $\text{NO}_3$ and $\text{PO}_4$ concentrations during storms

#### Management implications

→ Larger increases in stream  $\text{PO}_4$  and  $\text{NO}_3$  concentrations during storms in more highly disturbed catchments indicate that either: (1) sources are enhanced ( $\text{PO}_4$ , primary source is likely soils/sediments) or (2) sinks are reduced ( $\text{NO}_3$ , primary source is likely atmospheric deposition) by disturbance.

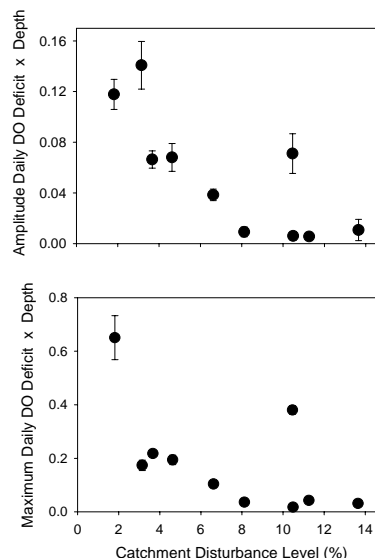
→ Larger storm increases in  $\text{PO}_4$  and  $\text{NO}_3$  concentrations will increase fluxes of these important nutrients off the base to the Chattahoochee River which could be a water quality concern. However, peak storm concentrations are still low relative to those found in most urban and agricultural drainages.

→ There was no evidence of a threshold disturbance level in the relationships between storm increases in  $\text{PO}_4$  and  $\text{NO}_3$  concentrations and disturbance level.



## Metabolism - Diurnal changes in stream dissolved oxygen concentrations

- **Negative relationships between the amplitude and maximum value of the DO saturation deficit and catchment disturbance level**
- **Indicates potentially reduced algal growth and microbial metabolism due to highly unstable and organic-poor sediments in streams in highly disturbed catchments**



Amplitude of the diurnal pattern in dissolved oxygen (DO) saturation deficit adjusted for water depth is an indicator of daily gross primary production rate. Maximum daily DO saturation deficit adjusted for water depth is an indicator of respiration rate. These metrics can also be indicators of disturbance if disturbance affects stream metabolism. There were negative relationships between the amplitude and maximum value of the DO saturation deficit and catchment disturbance level. We believe these are the result of reduced algal growth and microbial metabolism due to highly unstable and organic-poor sediments in streams in highly disturbed catchments.

### Measurement and calculations:

- (1) DO concentrations and water temperatures measured and recorded at 15-min intervals using in situ sondes (e.g., YSI 6000 series)
- (2) Calculation of DO saturation deficit using DO, water temperature, and barometric pressure data (see Benson and Krause 1980).



## Catchment-Scale Indicators

### Metabolism - Diurnal changes in stream dissolved oxygen concentrations

#### Management implications

→ Metrics characterizing diurnal changes in stream dissolved oxygen saturation deficits provide a good index of whole-system rates of gross primary production (GPP) and respiration (R).

→ Lower rates of stream GPP and R with catchment disturbance are the result of instability of streambeds and burial of organic substrata with organic-poor sands and clays, reducing habitat for algal and microbial colonization. Lower GPP and R will have negative effects on the productivity of stream macroinvertebrates and fish.

→ Stream GPP and R were very low in catchments with disturbance levels > 8%.



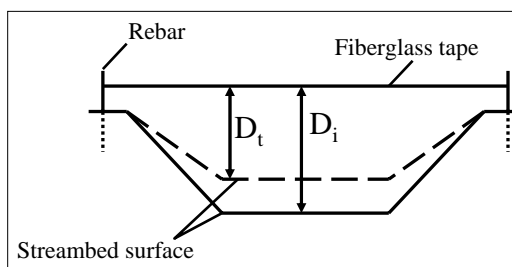
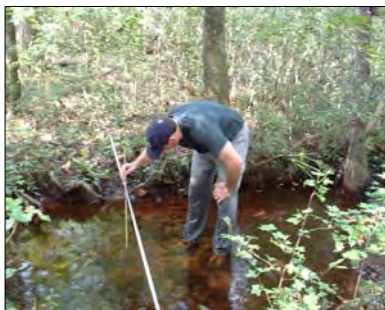
## Streambed Stability

Stability transects, leveled at deployment dates, measures taken every ~ 2 months.

Stability calculated as

$$\frac{\sum_{i=1}^n \sum_{z=1}^z |D_{iz,t} - D_{iz,i}|}{n}$$

where  $z$  is the distance along the transect,  $n$  is the number of transects in a stream and  $D$  is depth at time  $i$  and  $t$ .



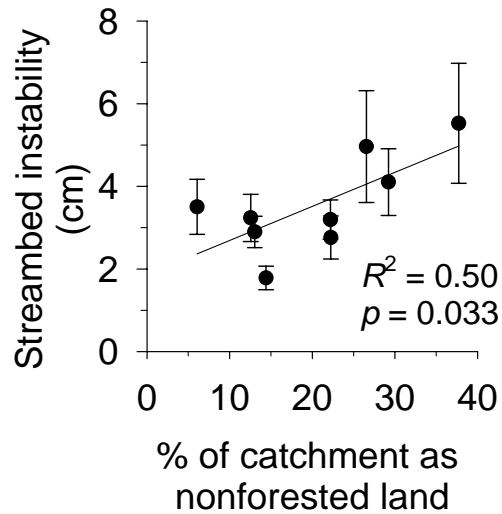
We estimated streambed instability by quantifying sediment movement using a modified transect method (Ray and Megahan 1979). We established cross-stream transects ( $n = 5$  per stream) by staking pairs of rebar on opposite banks of the channel perpendicular to flow. We leveled each transect (using a line level) and marked leveled heights on rebar pieces with cable ties. We quantified streambed height along fixed points of transects (20-cm intervals) by measuring vertical distance between the stream bottom and a fiberglass tape stretched across the channel; measures were made initially in January 2003 and then in July 2003 (~7 mo interval). Several storm events occurred during this sampling interval, so we considered this period sufficient to characterize relative changes in bed height among streams. We calculated streambed instability as the average absolute difference in height for each transect over the sampling period.



## Catchment-Scale Indicators

### Biotic habitat - streambed stability

- Bed instability increased with increasing disturbance intensity (as % of non-forested land).
- Suggests higher rates of erosion and subsequent sedimentation of stream within higher disturbed catchments
- Indicates reduced available habitat in more highly disturbed catchment



Streambed stability measures the amount of vertical change in bed height between 2 time periods (an estimate of bedload movement). In our streams bed instability increased with increasing disturbance intensity (as % of non-forested land). We believe these results are due to higher rates of erosion and subsequent sedimentation of stream within higher disturbed catchments.



## Catchment-Scale Indicators

### Biotic habitat - streambed stability

#### Management implications

→ A increase in bed instability indicates more movement of sediment as well as a reduction in available habitat for aquatic biota

→ Unrelated to bare ground and unpaved roads however significant positive relationship with non-forested land on slopes > 3%. The proportion of non-forested land includes fields and early successional vegetation, which may include historically disturbed areas. The inverse relationship between stability and non-forested land may indicate a land use legacy.



## Coarse woody debris, benthic particulate organic matter, and bed particle size



We used sediment cores (PVC pipe, area = 2.01 cm<sup>2</sup>, 10-cm depth) to quantify proportion of benthic particulate organic matter (BPOM) and streambed particle size. We considered BPOM all organic matter material  $\leq 1.6$  cm diameter, and quantified BPOM at 3 sites per stream every 2 mo (August 2001 to May 2003) and streambed particle size every 4 mo (September 2001 to May 2002). For BPOM analysis, we took 3 cores from the stream thalweg, oven-dried each sample at 80°C for 24 to 48 h, and then weighed it. Samples were then ashed in a muffle furnace at 550°C for 3 h, cooled in a desiccator, and reweighed; % BPOM was determined as the difference between dry and ashed masses divided by total dry mass. For particle size analysis, we collected 2 cores per site, 1 in the thalweg and 1 near the stream margin. We combined cores within each site ( $n = 3$ ), removed organic matter and dispersed particles following the pipette method from a 10-g subsample (Gee and Bauder 1986). Particle sizes were then separated by dry sieving (2.0, 1.0, 0.5, 0.250, 0.125, 0.063 and <0.053 cm fractions), and mean weighted particle size for each stream was estimated by multiplying the mass of each fraction by the midpoint between sieve fractions and then dividing by the total sample weight. Particles >2 mm were removed prior to the dispersing process (see Gee and Bauder 1986). However, we estimated the % of the entire sample that was >2 mm prior to dispersion and used this value to estimate the % of sample that would have been >2 mm in the 10 g subsample. For particle sizes occurring between 2 to 5 mm diameter (<10% of total particles, KOM, unpublished data), we assigned a midpoint size of 3.5 mm and included them in mean weighted particle size calculations.

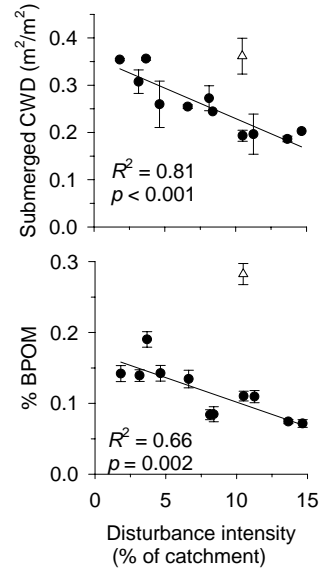
We quantified the relative abundance of coarse woody debris (CWD) in each stream during April 2002 and March 2003 using a modified transect method (see Wallace and Benke 1984). We quantified all submerged CWD >2.5 cm in diameter and all CWD buried within the upper 10 cm of the substrate along 15 cross-stream transects per stream; individual transects were 1-m wide with adjacent transects being spaced longitudinally 5-m apart. Live woody material (i.e., roots) was abundant in our study streams and appeared to be an important influence on channel structure, so we also included all live material in CWD measurements. CWD data were converted to planar area (m<sup>2</sup> of CWD per m<sup>2</sup> of stream bed) by multiplying CWD diameter by length and then dividing this value by the area sampled within each transect.



## Biotic habitat - coarse woody debris and benthic particulate organic matter

- Both CWD and BPOM decreased with catchment disturbance
- Suggests that with increasing disturbance a reduction in available habitat and base food resources occurs

Maloney, K. O., P.J. Mulholland, and J.W. Feminella. In press. The effects of catchment-scale military land use on stream physical and organic matter variables in small Southeastern Plains catchments (USA). Environmental Management.



Coarse woody debris (CWD) is an important bed stabilizer in sandy-bottomed streams and serves as important habitat for biota in sandy-bottomed streams. BPOM is a primary basal food resource in streams. The negative relationship between both CWD and BPOM with disturbance intensity suggests that with increasing disturbance a reduction in available habitat and base food resources occurs.



## Catchment-Scale Indicators

Biotic habitat - coarse woody debris and benthic particulate organic matter

### Management implications

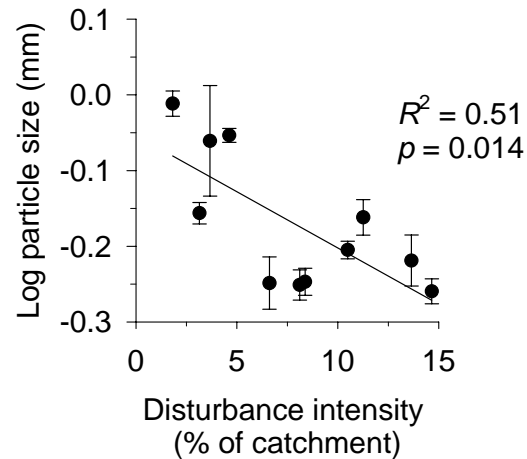
- A reduction in coarse woody debris and benthic organic matter signals a reduction in available habitat and basal food resources in these streams.
- Reduction in organic inputs as well as greater burial and transport downstream are likely explanations accounting for the lower CWD and BPOM levels in more disturbed catchments.
- Disturbance levels > 8-10% of catchment as bare ground and unpaved roads appeared to be a disturbance threshold for CWD and BPOM (consistent with that observed for several chemical patterns).



## Catchment-Scale Indicators

### Biotic habitat - sediment particle size

- Average bed particle size decreased with increasing catchment disturbance
- Suggests streams in higher disturbed catchment may have less available habitat for biota likely a result of increased sedimentation from the higher erosion rates associated with high disturbance



As the average particle size of a streambed decreases (especially in sandy streams) available habitat decreases. The inverse relationship observed between particle size and disturbance intensity suggests that streams in higher disturbed catchment may have less available habitat for biota likely a result of increased sedimentation from the higher erosion rates associated with high disturbance.



## Catchment-Scale Indicators

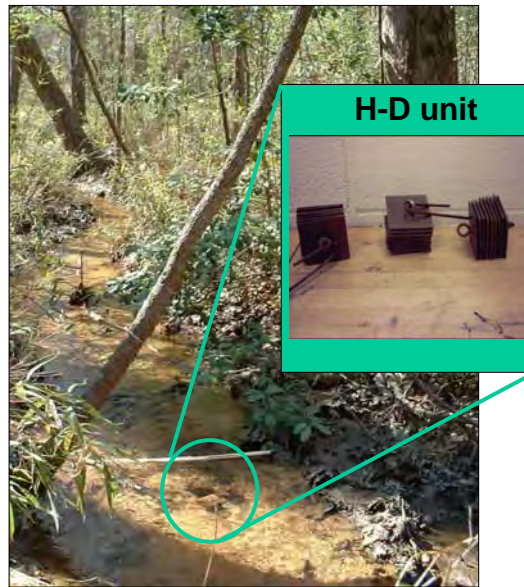
### Biotic habitat - sediment particle size

#### Management implications

- A reduction in bed stability indicates more movement of sediment as well as a reduction in available habitat for aquatic biota.
- Reduction in average particle size likely a function of the greater proportion of smaller, on-average, particles from eroded areas associated with catchment disturbance.
- Disturbance levels > 6.5% of catchment as bare ground and unpaved roads appeared to be a disturbance threshold for bed particle size (consistent with that observed for several chemical patterns).



## Macroinvertebrate assemblage

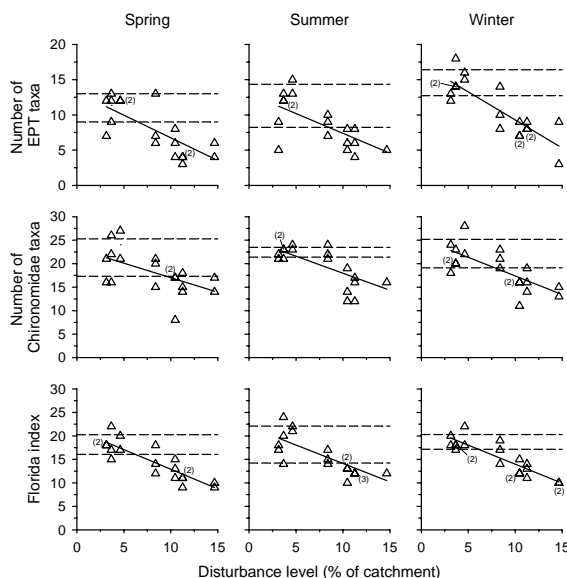


We quantified benthic macroinvertebrates in May (spring), September (summer), and January (winter) over a 3-y period (January 2000 to September 2002), using 4 Hester-Dendy (HD) multiplate sampling units (Merritt and Cummins, 1996), with 3 multiplates per unit, (total area = 0.33 m<sup>2</sup>). We left HDs in situ for a 6- to 8-wk colonization period. In addition, we also used 2 D-frame sweep net samples (250  $\mu$ m) to sample general benthic habitats (e.g., runs) in each study site, with net samples taken downstream and upstream of HD stations on each date. We field-preserved all samples in 95% EtOH, following elutriation of excess sediment. In the laboratory, we sorted the entire HD sample and subsampled sweep net samples (at least 200 organisms). We identified macroinvertebrates to the lowest taxonomic level possible (usually genus or morphospecies) using keys in Merritt and Cummins (1996), Wiggins (1996), and Epler (2001).



## Biota - Macroinvertebrates

- **Negative relationships between sensitive taxa (EPT), Number of Chironomidae taxa, and a regional defined tolerance index (Florida Index) with catchment disturbance level**
- **Suggests that with increasing disturbance benthic integrity decreases**



Benthic macroinvertebrates include multiple trophic levels and exhibit a vast array of sensitivity to disturbance making them ideal indicators of catchment disturbance. Regardless of season, observed negative relationships between sensitive taxa (EPT), Number of Chironomidae taxa, and a regional defined tolerance index (Florida Index) with catchment disturbance level. We believe these negative relationships are the result of reduced available habitat due to lower CWD and highly unstable and organic-poor sediments in streams in highly disturbed catchments.

### Measurement and calculations:

(1) EPT – number of aquatic insects in the orders Ephemeroptera, Plecoptera, and Trichoptera

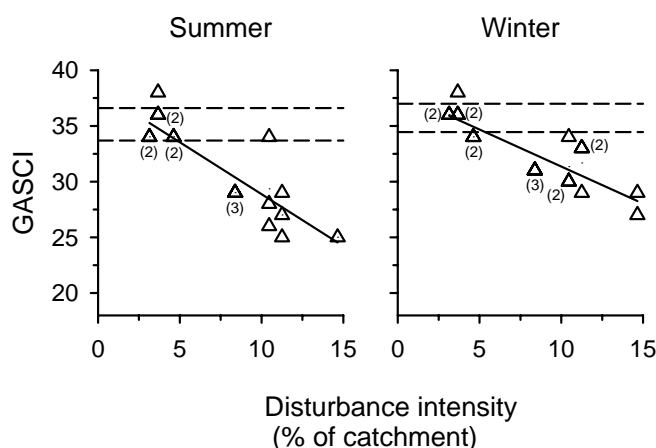
(2) Chironomidae taxa – number of taxa in the Dipteran family Chironomidae following keys in Epler (2001).

(3) The Florida index is a regionally defined tolerance index derived for Florida streams (see GADNR 2002). For this index, we separated taxa into 3 classes, with class 1 (sensitive) being assigned a value of 2, class 2 (moderately tolerant) assigned a value of 1, and class 3 (tolerant) a value of 0. The index is the sum of taxa in the respective classes, with lower and higher values indicating greater and lesser likelihood of stream impairment, respectively (FLDEP, 2002).



## Catchment-Scale Indicators Biota - Macroinvertebrates (GASCI)

- **Negative relationship between the GASCI with catchment disturbance level**
- **Suggests with increasing disturbance a reduction in biotic integrity occurs**



Maloney, K. O. and J. W. Feminella. In review. Benthic macroinvertebrate indicators of catchment disturbance at the Fort Benning Military Reservation, Ft. Benning, Georgia. Submitted to *Ecological Indicators*.

Benthic macroinvertebrates include multiple trophic levels and exhibit a vast array of sensitivity to disturbance making them ideal indicators of catchment disturbance. Regardless of season, observed a negative relationship between the GASCI with catchment disturbance level. We believe these relationships are the result of reduced available habitat due to lower CWD and highly unstable and organic-poor sediments in streams in highly disturbed catchments.

### Measurement and calculations:

The GASCI incorporates 7 macroinvertebrate metrics (taxa richness, EPT richness, Chironomidae richness, % dominant taxa, % Diptera, Florida Index, % Filterers) and 1 habitat metric (calculated by summation of assessment scores of 10 habitat characteristics, including channel sinuosity, bank stability, riparian vegetation features, see GADNR 2002). Each metric is assigned a score of 1, 3, or 5 depending on predefined ranges for each metric for the summer and winter seasons. For example, taxa richness has the following pre-defined ranges and scores: >30 taxa given a score of 5, 16–30 taxa score = 3, and <16 taxa score = 1. Each metric has predefined ranges with associated scores and the overall GASCI score is the summation of individual metric scores for each stream.



## Catchment-Scale Indicators

### Biota - Macroinvertebrates

#### Management implications

→ A reduction in sensitive taxa and lower tolerance index and multimetric scores with increasing catchment disturbance indicates a reduction in stream integrity with military training intensity. However even most disturbed sites were classified as “Good” using the multimetric index.

→ The reduction in sensitive taxa and lower tolerance index and multimetric scores with increasing catchment disturbance are likely a result of the altered water chemistry, increased flashiness, and reduced available habitat in the more disturbed catchments.

→ Disturbance levels > 8% of catchment as bare ground and unpaved roads appeared to be a disturbance threshold for reduced benthic macroinvertebrate integrity (consistent with that observed for several chemical patterns).



## Fish Assemblage

Electroshock  
Backpack



Photos of the 2 dominant fish collected *Semotilus thoreauianus* (left) and *Pteronotropis euryzonus* (right)



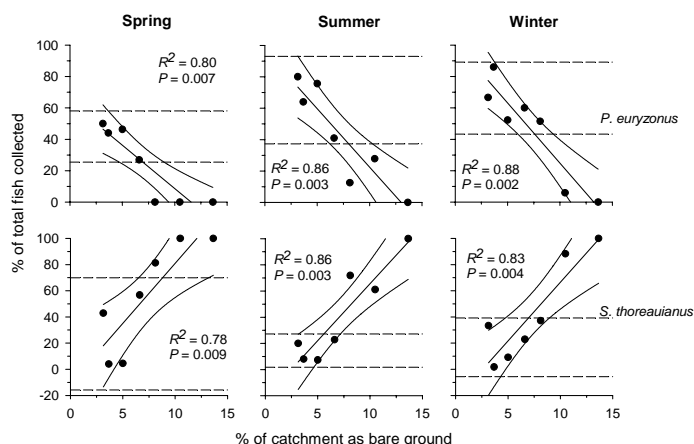
We sampled fish assemblages in 3 pools and 3 runs (riffles were not present) along a 100-m representative reach in each stream in March (spring), July (summer), and December (winter) 2003, following the 2-pass removal-depletion method using an electroshock backpack (Smith-Root LR-24) and block seines. We identified and recorded all fish collected and, except for voucher specimens used for taxonomic confirmation, we returned all individuals to the stream. We also recorded standard length (SL) for all fish collected, and quantified average depth and width (5 subsamples per unit) for each habitat (pool or run).



## Catchment-Scale Indicators

### Biota - Fish

- Proportion of *P. euryzonus* decreased with disturbance intensity, whereas the proportion as *S. thoreauianus* increased with disturbance intensity
- Suggests with increasing disturbance a reduction in biotic integrity occurs



Maloney, K. O., R. M. Mitchell, and J. W. Feminella. In prep. Influence of catchment disturbance on dominant fish assemblages in low-diversity headwater streams

Fish are often sensitive to disturbance and have a rich tradition in assessing stream integrity. Preliminary observations indicated that 2 fish species, *Pteronotopsis euryzonus* (broadstripe shiner) and *Semotilus thoreauianus* (Dixie chub), were numerically dominant in most of the study streams, therefore we used these species as focal populations to examine relationships between catchment disturbance and measures of individual populations. For these 2 numerically dominant fish species, we quantified the proportion of the total collected of both species for each stream and season. We used regression analysis to identify relationships between proportions of each of these species with catchment disturbance (as %BGRD). Regardless of season, the proportion of *P. euryzonus* decreased with disturbance intensity, whereas the proportion as *S. thoreauianus* increased with disturbance intensity. These opposing relationships suggest *P. euryzonus* is sensitive to disturbance resulting from military training, whereas *S. thoreauianus* is tolerant of such disturbance.

#### Measurement and calculations:

(1) The proportion of *S. thoreauianus* and *P. euryzonus* of the total collected per stream.



## Catchment-Scale Indicators

### Biota - Fish

#### Management implications

→ A reduction in sensitive taxa and increase in tolerant taxa with increasing catchment disturbance indicates a reduction in stream integrity with military training intensity. In fact, in the most disturbed catchment the sensitive taxa was not collected.

→ The opposite relationships with catchment disturbance are likely a result of different life history traits. *P. euryzonus* prefers deep flowing water with abundant CWD, are selective drift feeders, and require vegetation for spawning, whereas *S. thoreauianus* are omnivorous and deposit eggs into sediment. The culmination of increased SS, reduced CWD and bed stability associated with catchment disturbance likely affected *P. euryzonus* to a greater degree than *S. thoreauianus*.

→ Disturbance levels > 8% of catchment as bare ground and unpaved roads appeared to be a disturbance threshold for stream integrity using fish (consistent with that observed for several chemical patterns and macroinvertebrates).



## Research Conclusions

- Streamwater suspended solids were higher in more disturbed streams under both non-storm and storm flow periods.
- At baseflow streamwater concentrations of PO<sub>4</sub> and DOC declined with disturbance; however during storms concentrations of PO<sub>4</sub> and NO<sub>3</sub> increased to a greater extent in more disturbed catchments
- The daily amplitude and maximum deficit in DO concentrations were reduced in some seasons in more disturbed catchments.
- Reduced BPOM, CWD, bed particle size, and bed stability, concomitantly with increased stream flashiness with increasing catchment disturbance indicates less available habitat in more disturbed streams.
- Reduction in biotic integrity of streams with increasing disturbance.



## Ideal Indicators?

- Bare ground and unpaved road cover reflect tracked vehicle disturbance and general road use, dominant catchment-scale forms of disturbance at Fort Benning
- Stream suspended sediment concentrations (total and inorganic) during baseflow and storms are good indicators of catchment disturbance. Baseflow concentrations of PO<sub>4</sub> and DOC and stormflow concentrations of NO<sub>3</sub> and PO<sub>4</sub> are also good indicators of disturbance at catchment scales.
- Benthic macroinvertebrates and fish have wide range in tolerance and reflect cumulative responses to stress. Both macroinvertebrates and fish were good indicators.
- Habitat and biotic measures meet most of the criteria ideally associated with indicators (easily measured, sensitive, anticipatory, etc.)



## Photos of ISCO Sampler and YSI Sondes

ISCO model 6700 autosampler with model 750 area velocity module for measurement of water level and velocity. ISCO is deployed to collect 24 sequential water samples at regular intervals once a preset water level is reached



YSI model 6000 series sonde with DO and temperature/conductivity sensors deployed to record data at 15-minute intervals. Barometric pressure data used in calculations of DO saturation deficit are obtained from a nearby meteorological station and corrected to the stream site using elevation differences.





## **Technology Transfer of Ecological Indicators:**

Understory vegetation indicators of military  
disturbance in longleaf pine forests

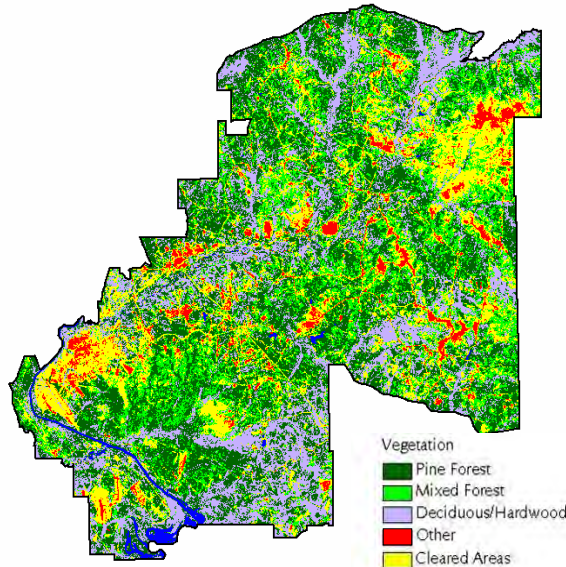


**Virginia Dale, Dan Druckenbrod and Suzanne Beyler,  
Principal Investigators  
Oak Ridge National Laboratory**



**75,000 ha land area**

**> 65,000 suitable  
 for longleaf pine  
 (*Pinus palustris*)**



Longleaf pine stands currently comprise approximately 5800 ha of the total area of Fort Benning. Soils favorable to the establishment and growth of longleaf pine make up approximately 65900 ha (about 90% of the total area).

Above is the 1995 Land Cover Classification of forest types at Fort Benning, Georgia produced by Oak Ridge National Laboratory.



## Ecological importance of longleaf pine forests:

- Federally-listed red-cockaded woodpecker (*Picoides borealis*)



Keystone species: Red Cockaded Woodpecker

***Only about 3% of original pine forest remains, endangering the Red Cockaded Woodpecker due to habitat loss.***



## Ecological importance of longleaf pine forests:

- Federally-listed red-cockaded woodpecker (*Picoides borealis*)
- Gopher tortoise (*Gopherus polyphemus*)





## Ecological importance of longleaf pine forests:

- Federally-listed red-cockaded woodpecker (*Picoides borealis*)
- Gopher tortoise (*Gopherus polyphemus*)
- Fire-adapted life-cycle of longleaf pine



Fire disturbance is natural and necessary feature of long leaf pine forests. Fire burns the decaying matter and competing tree species which then provides broken-down soil nutrients for the pines which normally grow in very poor (nutrient deficient) soils. The pines also have a fire-resistant bark and needles so that they can survive and adapt to the changes brought about by fire.



## Ecological importance of longleaf pine forests:

- Federally-listed red-cockaded woodpecker (*Picoides borealis*)
- Gopher tortoise (*Gopherus polyphemus*)
- Fire-adapted life-cycle of longleaf pine
- Understory serves central role in pathway of forest development



Understory as ecological indicator (Dale et al. 2002)



## Ecological importance of longleaf pine forests:

- Federally-listed red-cockaded woodpecker (*Picoides borealis*)
- Gopher tortoise (*Gopherus polyphemus*)
- Fire-adapted life-cycle of longleaf pine
- Understory serves central role in pathway of forest development
- Bracken fern (*Pteridium aquilinum*)



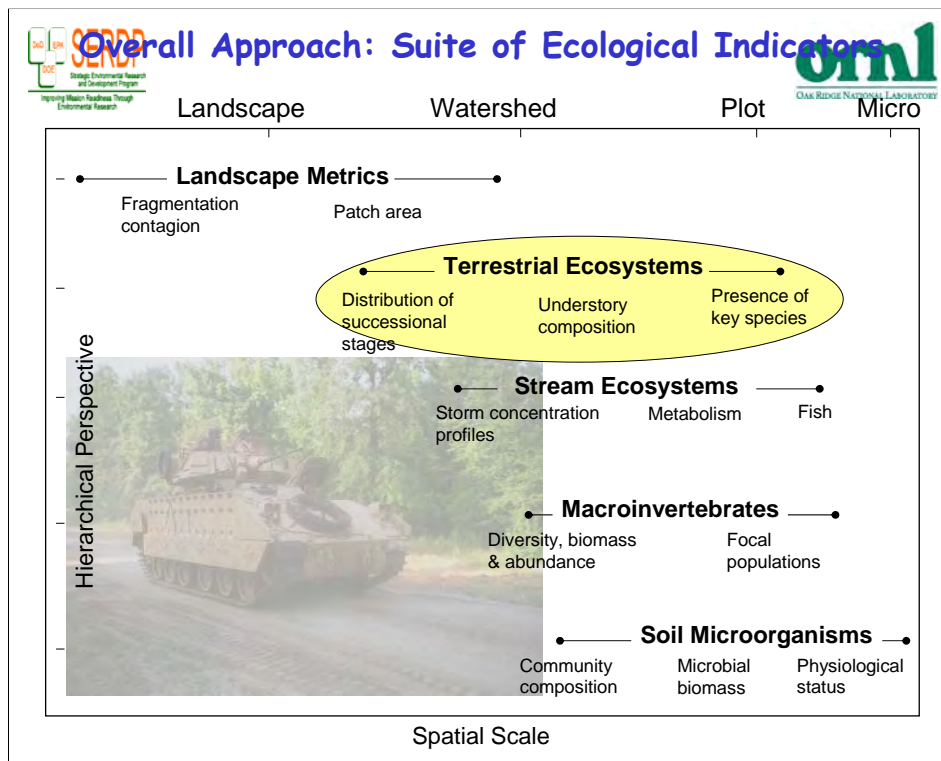


## Military importance of Fort Benning:



- Home of the Infantry
- Wide range of active training resulting in foot and tracked-vehicle movements
- Military and ecological objectives result in a dual charge for land managers
- Need for a 'good' indicator of understory

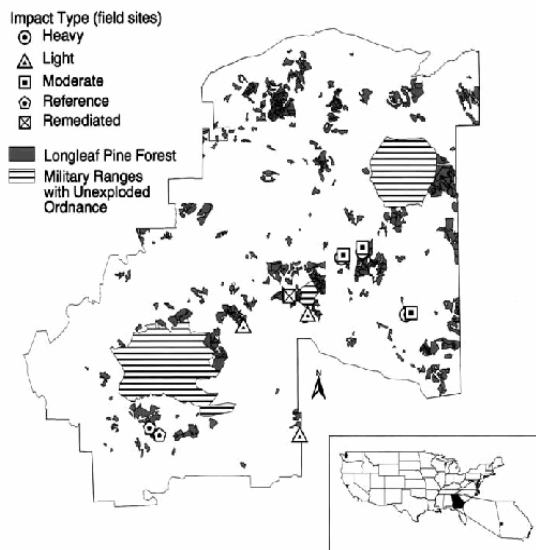




The response of understory vegetation is part of a larger project designed to investigate indicators that would be useful to augment current sampling regimes at military bases and typical of other actively managed sites. Current ecological monitoring on military lands, the Land Condition Trend Analysis (LCTA) (Diersing et al. 1992), does not incorporate the diversity of indicators that are necessary for monitoring changes and responses to land. We hypothesized that understory conditions are a key element in the suite of indicators that can reflect differences in military training intensity. While some of the indicators from the proposed suite are designed to measure changes that occur over the long term, understory vegetation is the element representing ecological changes that may occur over a few years to decades. Before such a suite can be adopted, it is necessary to evaluate how effectively the component indicators represent changes and susceptibility of ecological systems to military training. The purpose of these studies are to examine the ability of understory vegetation to indicate differences in disturbance regimes.



## Understory survey across a gradient of training intensity

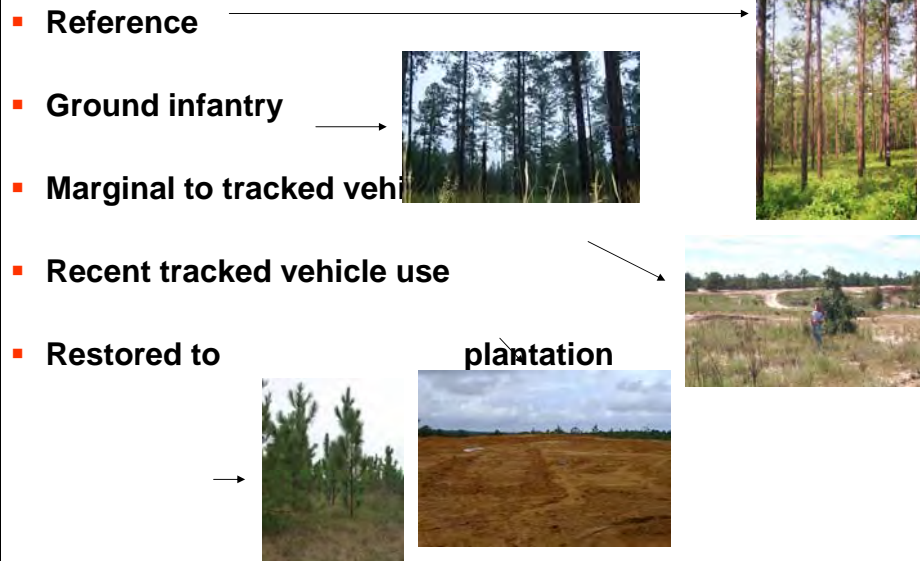


Dale, V.H., S.C. Beyeler, and B. Jackson. 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. *Ecological Indicators* 1: 155-170.

Study site locations were on land suitable for longleaf pine growth. Determination of potential site locations was achieved through a combination of existing forest stand information (Personal Communication, Bob Larimore 1999, Fort Benning, GA) and county soil surveys of the United States Department of Agriculture Natural Resource Conservation Service (USDA NRCS 1924, 1983, 1993, 1997). We overlaid an image of the United States Forest Service forest stand classification onto USDA NRCS soil maps for the area of land within the Fort Benning boundary. A final map was then created depicting locations of soils associated with longleaf pine within the installation boundary, and study sites were selected from those areas.



## Observational Study: land use types



The study was designed based on a stratified sampling methodology. The sampling sites were blocked into five training intensity categories: reference, light, moderate, heavy, and remediation. Reference areas experience little to no training activities and are often in exclusion zones around firing ranges. Light training areas are limited to infantry training and individual orienteering activities. Moderate training areas occur adjacent to tank training zones and are, thus, exposed to some tracked vehicle maneuvers, as well as limited vehicle and infantry traffic. Heavy training areas are used exclusively for wheeled and tracked vehicle training exercises. The classification of each site was primarily based on historical records of training activity; however, due to the variability of training intensity over space, final site selection was achieved through field reconnaissance and discussions with the Fort Benning natural resource personnel.



## Observational Study: Methods

|   | Species cover<br>scale             | Percentage<br>Midpoints |
|---|------------------------------------|-------------------------|
| <ul style="list-style-type: none"> <li>Established 5m circular plots along transects in each training category</li> </ul> | No plants present                  | 0                       |
|   | < 1% cover, 1-5 small individuals  | 0.25                    |
| <ul style="list-style-type: none"> <li>Understory surveyed with Braun-Blanquet percent cover scale in 1999</li> </ul>     | < 1% cover, many small individuals | 0.5                     |
|   | < 1% cover, few large individuals  | 0.75                    |
|   | 1-5% cover                         | 3.0                     |
|   | 5-12% cover                        | 8.5                     |
|   | 12-25% cover                       | 18.5                    |
|   | 25-50% cover                       | 37.5                    |
|   | 50-75% cover                       | 62.5                    |
|   | 75-100% cover                      | 87.5                    |

Five circular plots were established along each transect at intervals of 15 m between the centers. The circular plot size of 5 m radius was determined based upon a species-area curve constructed for the reference site. At that size plot, 31 understory species occurred. Within each plot, all species of understory vegetation were identified and assigned a cover class using a modified Braun-Blanquet (1932) cover system (based on Clarke 1986 p. 64). Bråkenhielm and Qinghong (1995) have demonstrated that visual estimates provide the most accurate, sensitive, and precise measure of vegetation cover compared to point frequency and subplot frequency methods. Thus, visual estimates of understory cover were used in this study. The field crew was calibrated as to the appearance of each cover class. Individual species cover scores could not exceed 100%; however cumulative cover scores for all species associated with an individual plot could be larger than 100%.



## Observational Study: Results

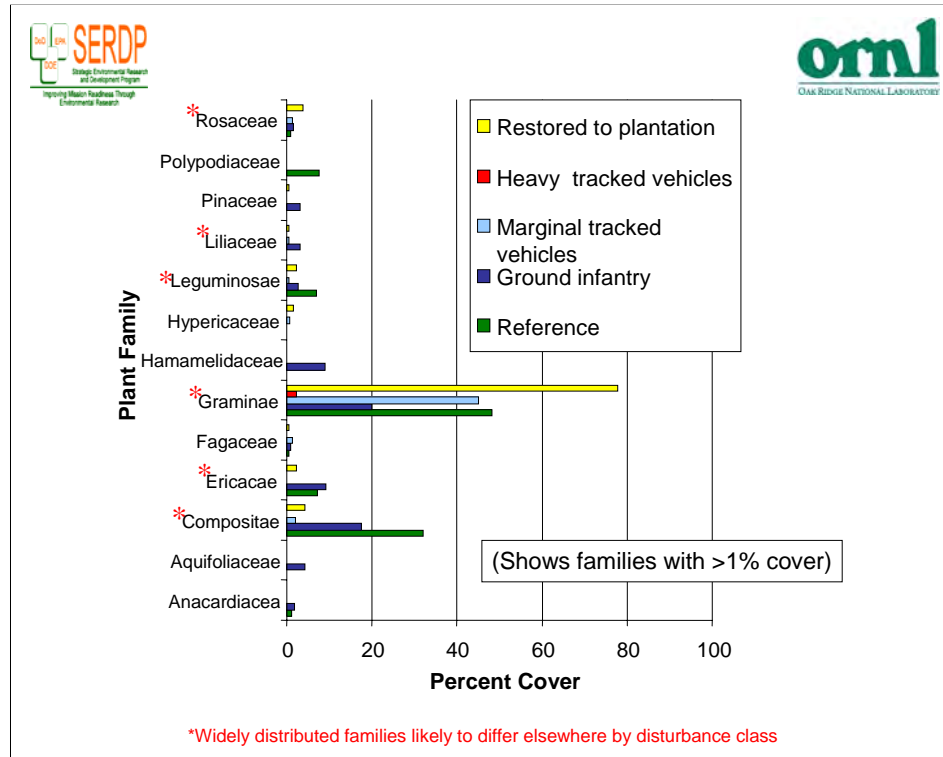
- Understory survey across a gradient of training intensity
- No clear pattern in total understory cover across gradient
- High variability in species richness across gradient

| Training Category        | Percent Understory Cover | Understory Species Richness |
|--------------------------|--------------------------|-----------------------------|
| Reference                | <b>67.0 (10.8)</b>       | <b>82</b>                   |
| Ground Infantry          | <b>67.9 (12.9)</b>       | <b>95</b>                   |
| Marginal Tracked Vehicle | <b>44.4 (18.2)</b>       | <b>78</b>                   |
| Recent Tracked Vehicle   | <b>4.7 (4.5)</b>         | <b>16</b>                   |
| Remediation              | <b>67.0 (17.2)</b>       | <b>69</b>                   |

Except for heavy training areas, these data suggest that neither understory cover nor plant diversity are useful indicators of past training.

The high diversity and large variation in understory cover of longleaf pine forests and reestablishing vegetation provided a challenge in the use of understory species to distinguish between training impacts in longleaf pine stands. It was not surprising that little blue stem (*Andropogon scoparius*) contributed the highest mean cover over all sites, for it is a characteristic plant of longleaf pine forests. Species that were only identified from one type of training area sometimes were helpful in identifying characteristics of such sites. For example, bracken fern (*Pteridium aquilinum*) was only found in reference sites and is a typical plant of old growth longleaf pine stands. Prickly pear (*Opuntia compressa*) was only found in moderately disturbed sites and can likely withstand the stressful conditions of such sites. The high variability in understory vegetation cover over training categories probably led to the lack of separation by training category by species which required analyses based upon groupings of species into life forms and families.





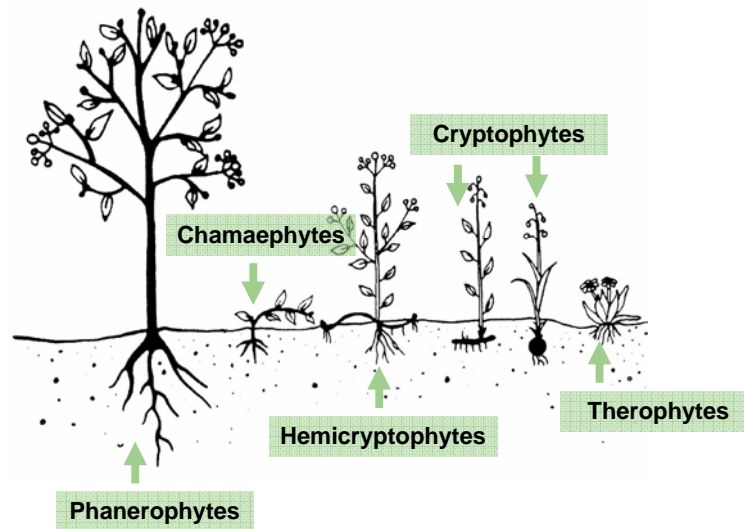
Plant families are also a useful way to group understory vegetation to reflect differences in training regimes. Of those families that contribute more than one percent cover, light training areas had the highest diversity with 11 families represented whereas heavy training areas had only one family present. Anacardiaceae was the most abundant family in the light training sites (possible because foot soldiers may have avoided poison ivy, one of the common representatives of this family, giving it a competitive advantage over other species that were more readily tramped upon). Both remediated and reference sites each contained six families with greater than one percent cover, but three of these families were not the same. Ferns (Polypodiaceae) and, in particular, bracken fern (*Pteridium aquilinum*) were distinct to reference sites and can be assumed to be an indicator of the absence of military disturbance.

Graminae was the only family common to all training types. Yorks et al. (1997) report from their literature review of foot traffic impacts that graminoids were found to be most resistant. Grasses contributed very little cover in the heavily trained sites but provided more than 70% cover to the remediated sites. It is not surprising that remediated sites had such high cover of grasses, for recovery efforts of these areas included planting grass seed. The relevant management question is: does such planting bring impacted sites closer to the vegetative characteristic of naturally revegetating sites? We found no family that was distinct to remediated sites. Except for the low percentage of trees and shrubs, life form distribution of remediated sites is similar to that of both reference and lightly trained sites with cryptophytes being well represented. Thus, this analysis suggests that the remediated sites are moving along the pathway toward established vegetation much like that of the reference or lightly trained areas.



## Raunkiaer Life Forms

Classifies plants into functional types



All species were also classified using Raunkiaer's life form classification system (Kershaw and Looney 1985) based on the height of perennating buds.

Phanerophytes - trees or shrubs with buds greater than 0.5 m above ground

Chamaephytes - plants with buds that are 0.1 to 0.5 above ground

Hemicryptophytes - plants with buds that at the ground surface

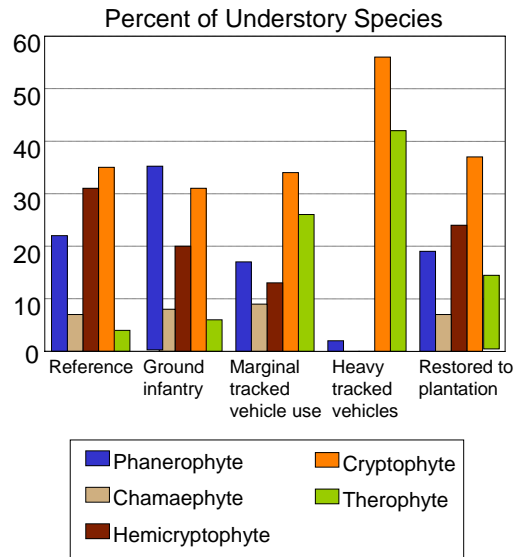
Cryptophytes - plants with below-ground dormant tissue

Therophytes - annuals



## Observational Study: Results

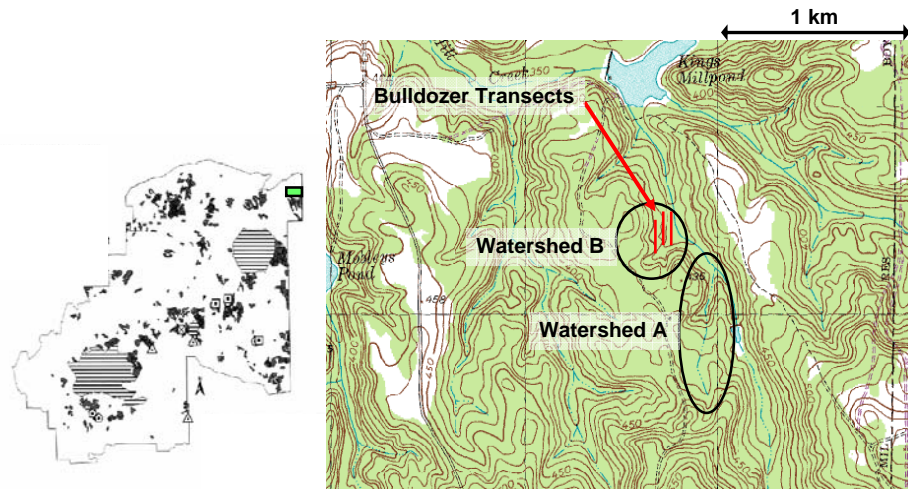
- Functional groupings of species displayed consistent trends across gradient
- Phanerophytes increase under infantry, but decrease under recent tracked vehicle
- Cryptophytes increase under recent tracked vehicle
- Therophytes increase under any use



In contrast to considering diversity and cover of all species, life form offers a more effective indicator of past disturbances. Frequency of life form occurrence distinguishes between military disturbances. Trees and shrubs (Phanerophytes), which may be less affected by foot soldier traffic than other life forms, dominate cover in light training areas. However, in an extensive literature review of foot traffic impacts on vegetation, shrubs and trees suffered the longest lasting decrease (Yorks et al. 1997). Our analysis suggests that foot traffic impact on trees and shrubs may not be as intense as on other life forms. This difference between duration and intensity of disturbance impacts is a necessary distinction (White and Pickett 1985). Cryptophytes dominate in all other training categories possibly because they are common in the flora due to their ability to withstand ground fires, the natural disturbance of longleaf pine forests. Plants with underground buds are possibly the only vegetation able to withstand heavy tank traffic. In contrast, therophytes, which are also found in the heavy training areas, likely seeded into sites after tanks training ceased. Chamaephytes do not contribute more than one percent cover for any training treatment possibly because they are uncommon in the long leaf pine flora and because they are susceptible to all types of traffic.



## Experimental Study



This experimental study was set with a context of a paired watershed study in K11. These watersheds encompass several research studies, but this section of the report focuses on the response of understory vegetation to tracked-vehicle disturbance.



## Experimental Study: Methods

- **Tracked-vehicle created 3 disturbance transects using D7 bulldozer in May 2003**
- **Established 10 treatment and 10 control plots per transect for total of 60 plots**
- **Sites previously treated with prescribed-burning and selectively-thinning in 2002**



Druckenbrod, D.L. and V.H. Dale. In prep. Experimental response of understory plants to mechanized disturbance in longleaf pine forests. *Ecological Applications*.

This study was set within longleaf pine habitat in the K-11 training area at Fort Benning, Georgia. In May 2003, soil disturbance was generated through the movement of a D7 bulldozer with its blade lowered to create three, 50 m treatment transects. This treatment not only approximated the movement of mechanized vehicles often utilized in training activities at Fort Benning, but also constituted a more uncommon and severe disturbance than the more frequent fire-induced disturbances in longleaf pine ecosystems. The bulldozer removed most vegetation and organic matter from the soil surface exposing a sandy substrate. A paired control transect was established 5 m adjacent to each disturbance treatment transect. The three paired transects were located within sites previously manipulated by fire and thinning treatments in 2002 and were generally parallel with elevation contour lines in the watershed.



## Experimental Study: Methods

|  | Species cover<br>scale                | Percentage<br>Midpoints |
|--|---------------------------------------|-------------------------|
| Understory surveyed with Braun-<br>Blanquet percent cover scale in June<br>2003, Sept 2003, June 2004, and<br>Oct 2004 | No plants present                     | 0                       |
|  | < 1% cover, 1-5 small<br>individuals  | 0.25                    |
|  | < 1% cover, many<br>small individuals | 0.5                     |
|  | < 1% cover, few large<br>individuals  | 0.75                    |
|  | 1-5% cover                            | 3.0                     |
|  | 5-12% cover                           | 8.5                     |
|  | 12-25% cover                          | 18.5                    |
|  | 25-50% cover                          | 37.5                    |
|  | 50-75% cover                          | 62.5                    |
|  | 75-100% cover                         | 87.5                    |

Along each transect, 10 circular plots were chosen at random distances. Each plot encompassed 0.568 m<sup>2</sup>. Within each plot, all vegetation less than 1 m in height was surveyed using a modified form of the Braun-Blanquet (1932) cover system (Clarke 1986, Dale et al. 2002) (Table 1). Surveys were conducted a total of four times over two growing seasons: June 2003, September 2003, June 2004, and October 2004. Cover classes were assigned to all species present, total plant cover, bare ground, and leaf litter.

Species identification relied on Radford et al. (1968), Godfrey (1988), and Weakley (2004). Species identifications were confirmed with voucher specimens at the University of Tennessee Herbarium and species nomenclature follows the Integrated Taxonomic Information System (U.S. Department of Agriculture 2004a).



## Experimental Study: Methods

- Raunkiaer (1934) Life-forms used as Functional Types
- % cover transformed using arcsine square-root
- Repeated-Measures ANOVA with SPSS 12.0 GLM



Consistent with Dale et al. (2002), functional classifications for all vascular taxa were delimited using Raunkiaer's (1934) life-form classification system, which delineates life forms by the height of perennating buds (see also Kershaw and Looney 1985). While Raunkiaer envisioned life forms as a response of plants to unfavorable climate, his approach has since been also successfully applied to the response of plants from disturbance (McIntyre et al. 1995). Classification of species into life forms was assisted by life-history information from Radford et al. (1968), The National PLANTS Database (U.S. Department of Agriculture, Natural Resources Conservation Service 2004), and Fire Effects Information System (U.S. Department of Agriculture 2004b). Cover values were assigned for each life form by plot.

Data were analyzed using Matlab (Mathworks, 2004). Cover-abundance values of total plant, litter, bare ground, and Raunkiaer life forms were converted to percentage midpoints and summed by life form per plot. An arcsine square root transformation for proportional data was employed to approximate a normal distribution (Zar 1996). Species and family richness counts were log<sub>10</sub> transformed. Repeated measures analysis of variance (ANOVAR) was employed to compare cover values between treatment and control plots and also to analyze trends in cover through time using orthogonal polynomial contrasts (Keppel 1991). ANOVAR has been widely applied to the analysis of ecological trends through time (see Gurevitch and Chester 1986, Moser et al. 1990, Potvin et al. 1990, Meredith and Stehman 1991, Nuzzo et al. 1996). Orthogonal polynomials including linear, quadratic, and cubic trends were considered in the response of understory plant growth.



## Experimental Study: Methods

**Examined 10 Possible Indicators of Understory Disturbance from Mechanized Vehicles:**

- **Total Plant Cover**
- **Bare Ground Cover**
- **Leaf Litter Cover**
- **Species Richness**
- **Family Richness**
- **Therophytes**
- **Cryptophytes**
- **Hemicryptophytes**
- **Chamaephytes**
- **Phanerophytes**



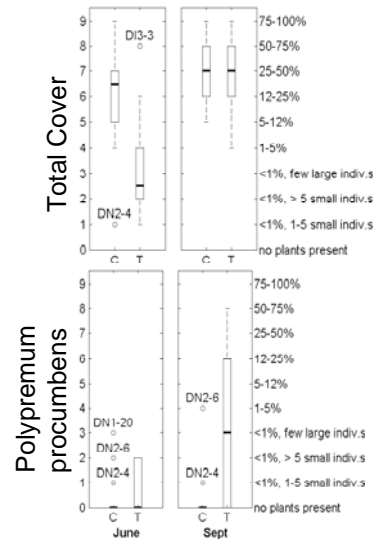


Visual response of disturbance transects through time showing return of vegetative cover over two growing seasons.



## Experimental Study: Results

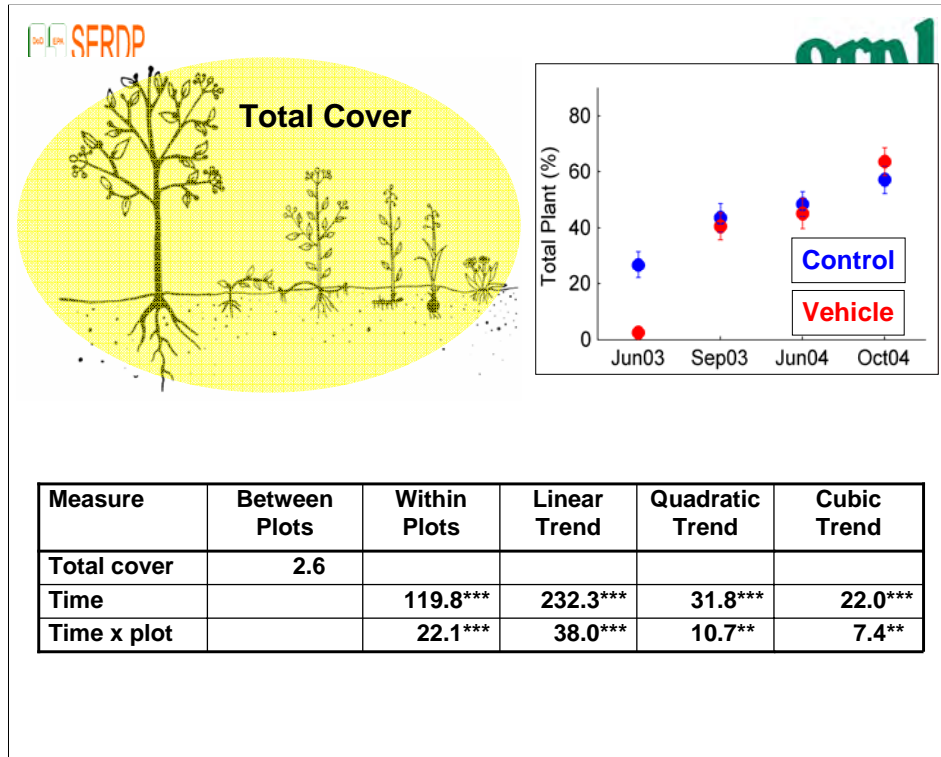
- Results after 1 growing season
- “Typical” measures of understory plant communities:
  - Total Cover
  - Indicator Species
- What is the long-term functional response of vegetation (i.e. by Raunkiaer life forms)?



Dale, V.H., D.L. Druckenbrod, L. Baskaran, M. Aldridge, M. Berry, C. Garten, L. Olsen, R. Efroymson, and R. Washington-Allen. In press. Vehicle impacts on the environment at different spatial scales. *Journal of Terramechanics*.

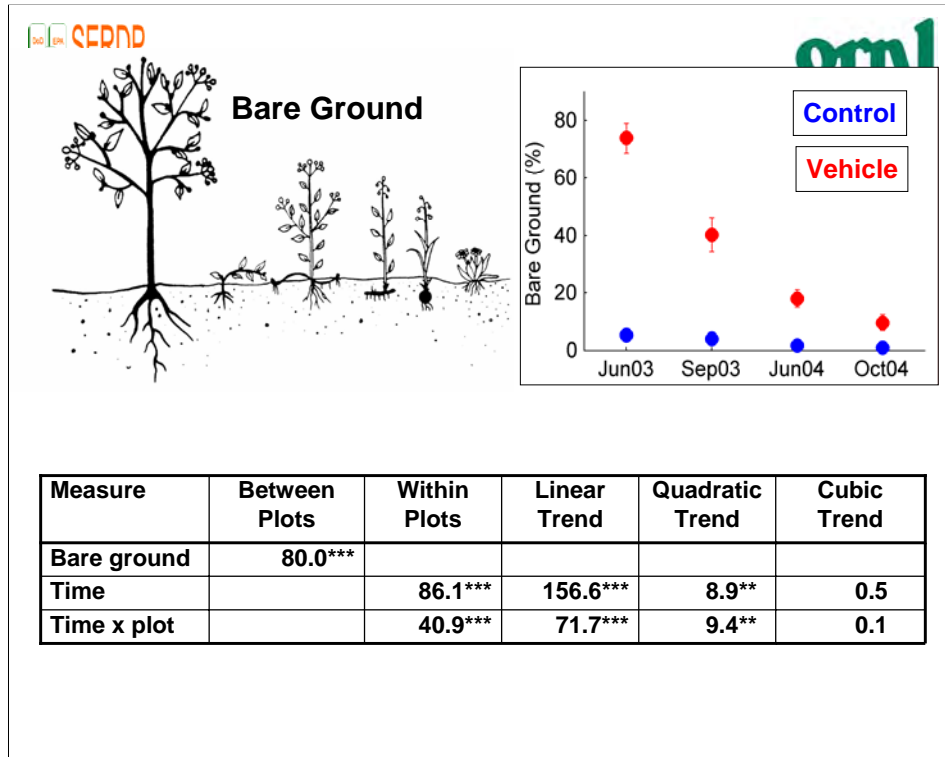
The response of total plant cover to mechanized disturbance shows a remarkable recovery by 4 months after the disturbance to an equivalent value of the control transects. The rapid recovery occurred over the growing season and during an abnormally wet summer, even though soil compaction was certainly of longer duration. Despite the renewed cover, however, vegetation composition became significantly different from that in the control transects. The September survey showed a significant increase in juniper leaf (*Polypremum procumbens*). This species-specific increase agrees with the species ecology described by Radford et al. (1968), who notes that *P. procumbens* is found within habitats showing recent disturbance, including roadsides.





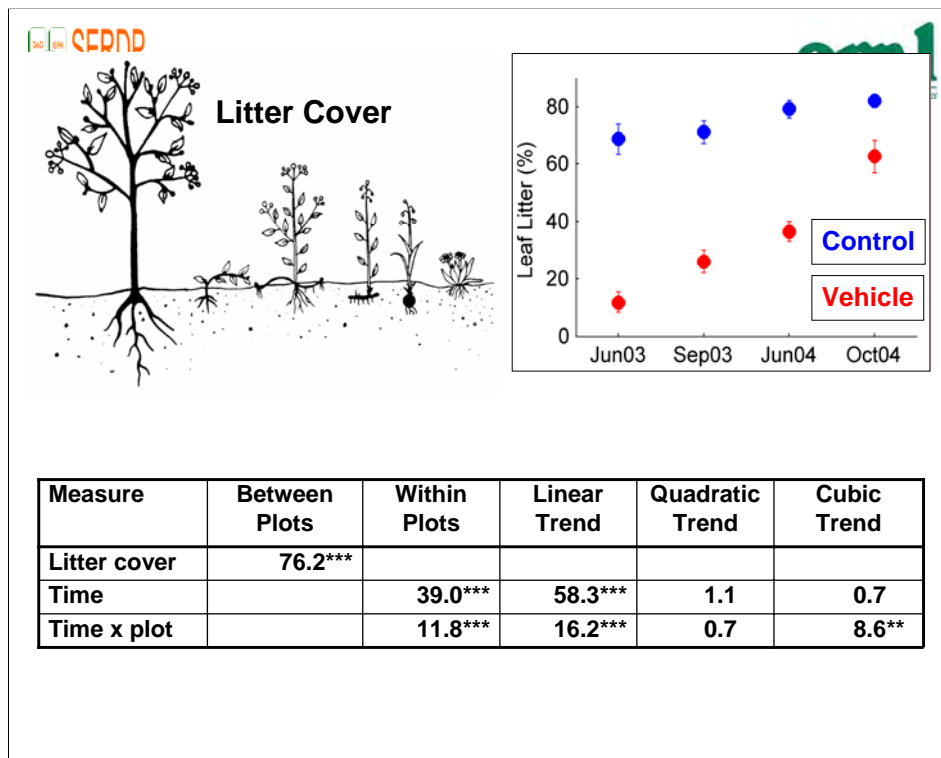
Total plant cover linearly increased through time when averaged across control and treatment plots, but treatment plots linearly increased their cover at a greater rate than control plots. Thus, treatment plots return to control plot covers by end of first growing season.





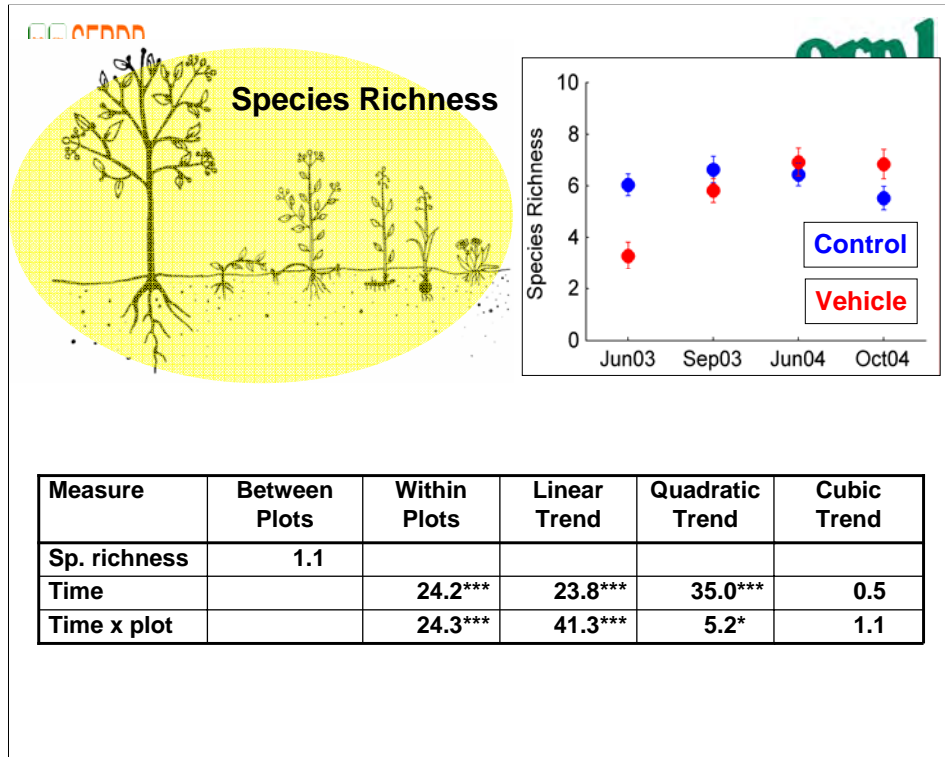
Bare ground cover linearly decreased when averaged across control and treatment plots, but this reduction was greater for the treatment plots. Treatment plots converge on control values after two years.





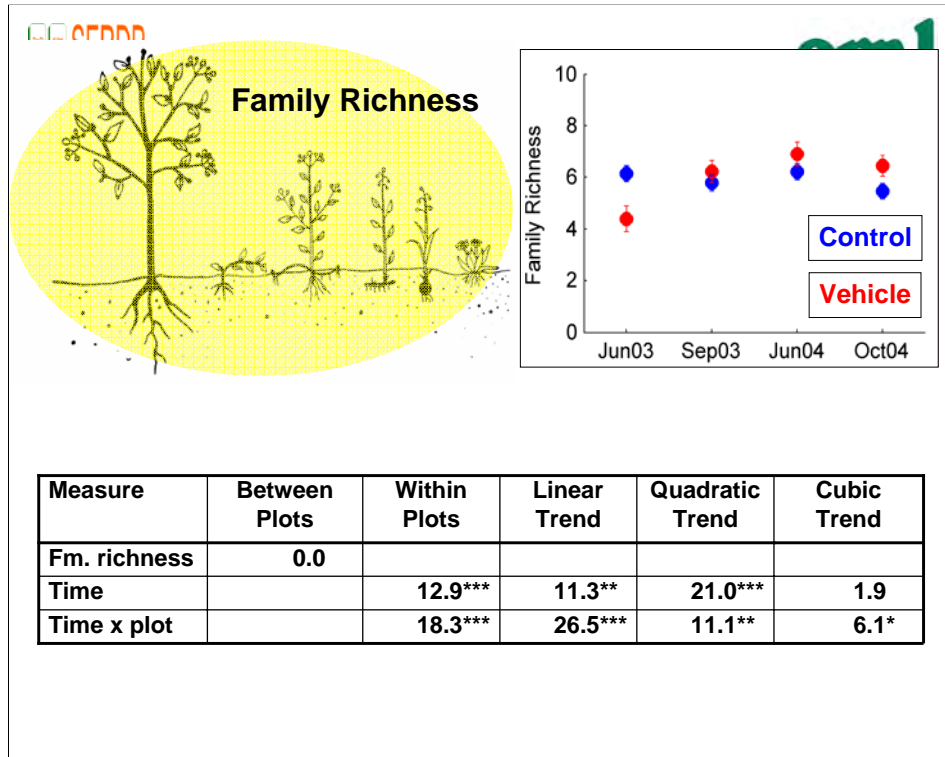
Litter cover linearly increased through time across all plots with a greater linear increase observed in the treatment plots through time. So, litter cover also converges on control values after 2 years.





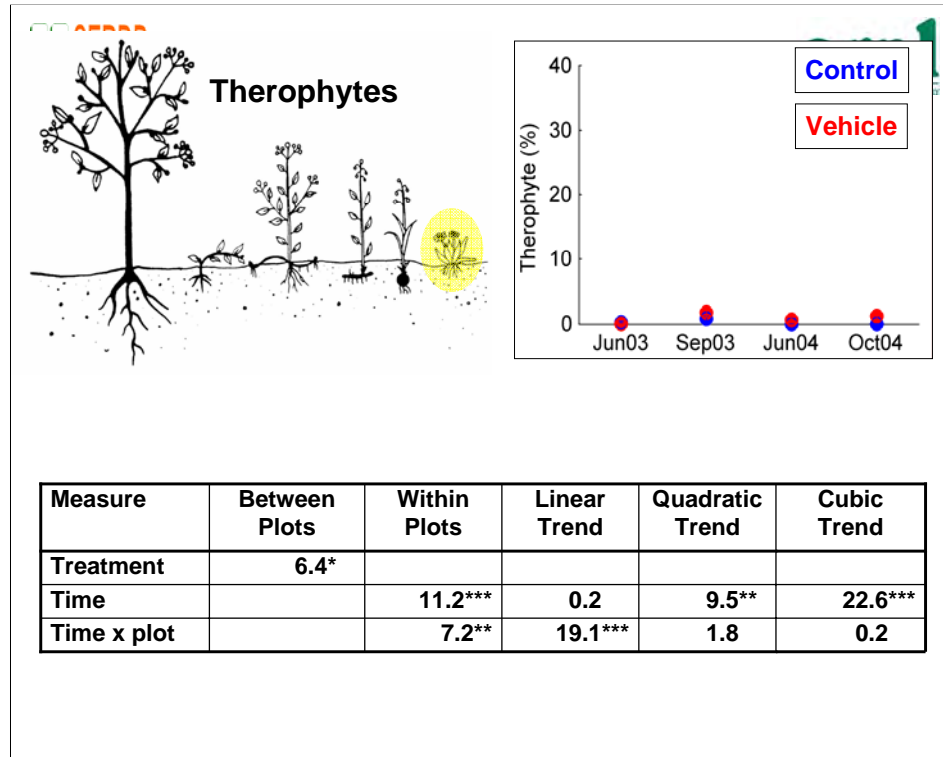
Species richness responded showing a linear increase over time across all plots with the treatment plots increasingly linearly at a greater rate than the control plots. This result suggests that species richness may eventually be promoted by mechanized disturbance.





Family richness also responded showing a linear increase over time across all plots with the treatment plots increasingly linearly at a greater rate than the control plots. This result suggests that family richness may also eventually be promoted by mechanized disturbance.



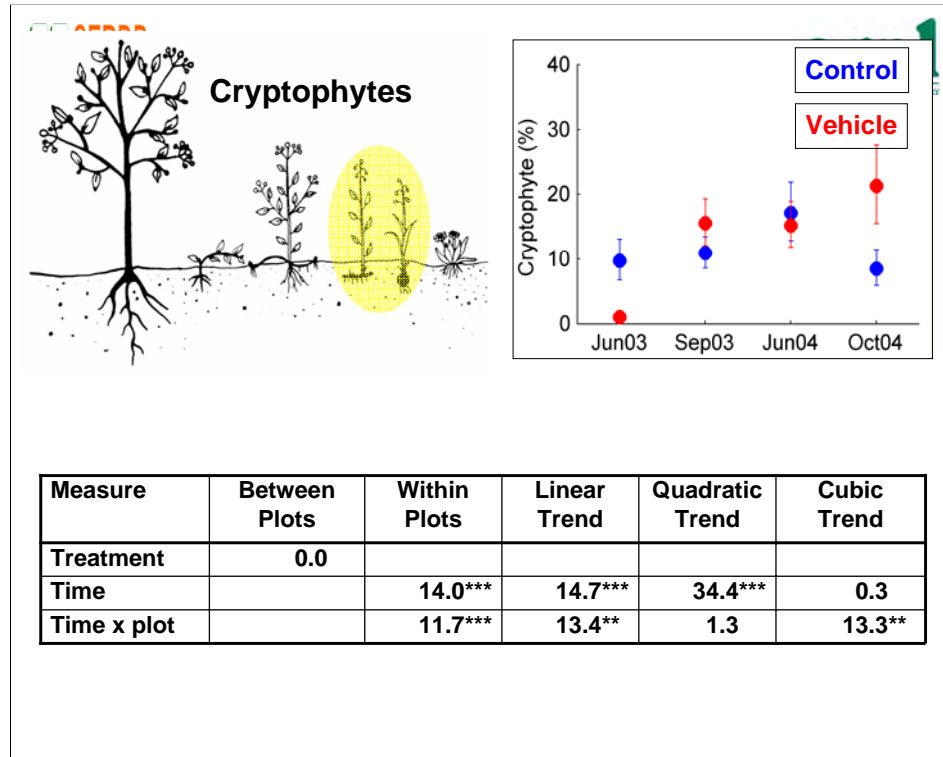


Therophytes showed a significant disturbance treatment effect, but are the least prevalent life form cover.

Therophytes did not show a linear trend across all plots, but did show a linear interaction of disturbance treatment with time.

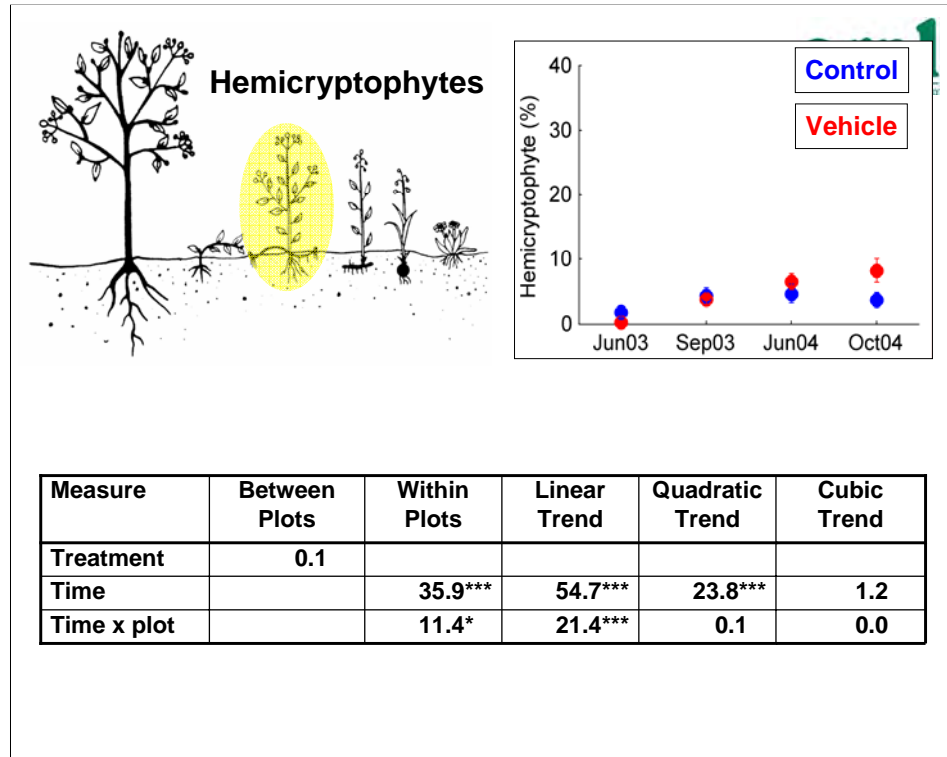
Chamaephytes, Hemicryptophytes, Cryptophytes, and Therophytes also all increased following a quadratic trend across all plots; however, this model explained the trend better than the linear model only for Cryptophytes and Therophytes. Cubic trends were only evident in the interaction of disturbance treatment with time for Cryptophytes and the trend across all plots for Therophytes.





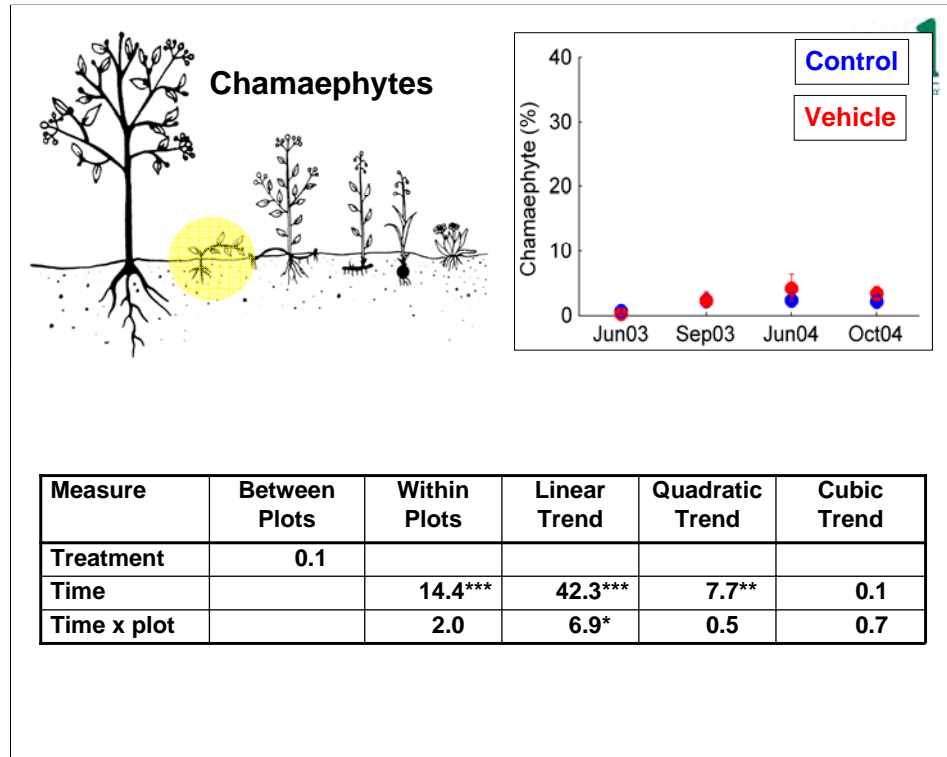
Cryptophytes also showed a significant effect of time and interaction of time with treatment. Cryptophytes increased linearly across all plots through the two growing seasons with treatment plots displaying a significantly greater linear rate of increase than control plots. This result suggests that Cryptophytes may eventually be promoted by mechanized disturbance.





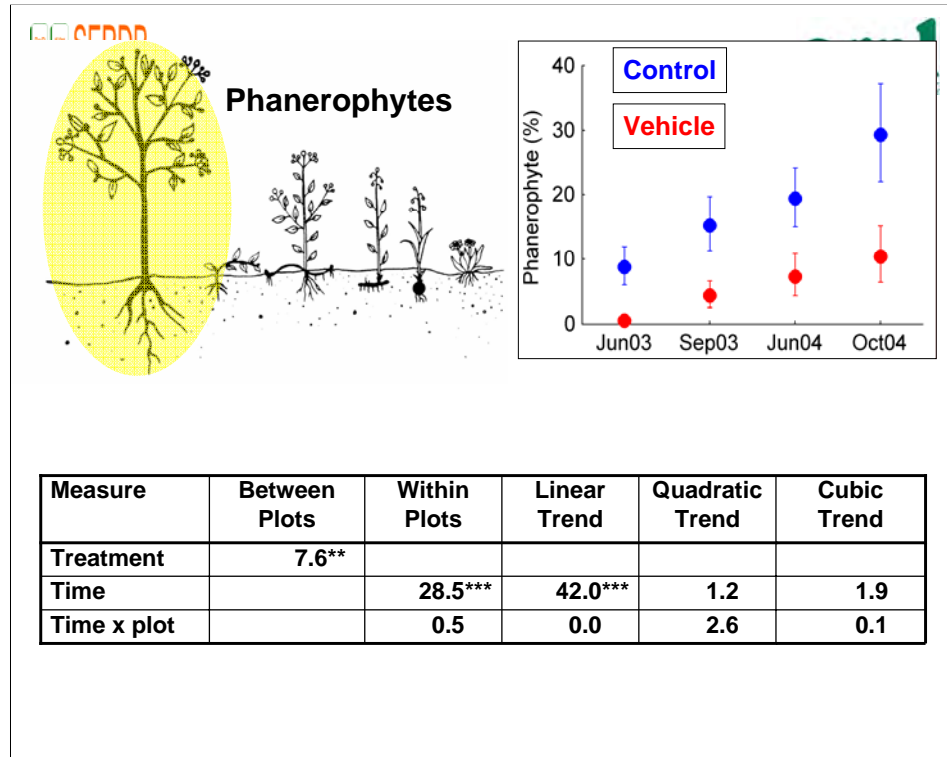
Hemicryptophytes showed a significant effect of time and interaction of time with treatment. Hemicryptophytes increased linearly across all plots through the two growing seasons with treatment plots displaying a significantly greater linear rate of increase than control plots. This result suggests that Hemicryptophytes may eventually be promoted by mechanized disturbance.





Chamaephytes did not show a treatment effect and only a slightly significant linear interaction trend with disturbance treatment as treatment plots increased a greater rate. Chamaephytes do not appear to be good indicators of disturbance.





Phanerophytes showed a significant disturbance treatment effect over the two growing seasons. Phanerophytes also displayed a time effect, with an increasing linear trend, but no interaction of treatment with time as a main effect. Thus, Phanerophytes show no indication of recovery after 2 years.



## Research Conclusions

- **Total cover initially reduced; however, returns to control levels within growing season**
- **Phanerophytes and Cryptophytes account for majority of understory cover for both control and treatment**
- **Phanerophytes display reduced cover through time**
- **Chamaephytes increase cover through time but not strongly impacted by tracked-vehicle disturbance**
- **Hemicryptophyte, Cryptophyte, and Therophyte cover may be promoted through time by tracked-vehicle disturbance**



## Ideal Indicators?

- Bare ground and litter cover reflect tracked vehicle disturbance, but total cover masks changes to vegetation
- Family richness and functional types reflect change in vegetation composition and structure
- Phanerophytes readily serve as an indicator of understory structure and cover that are reduced by tracked-vehicle activity
- Hemicryptophytes, Cryptophytes, and Therophytes may also serve as an indicator that are promoted by tracked-vehicle activity
- These functional types meet most of the criteria ideally associated with indicators (easily measured, sensitive, anticipatory, etc.)



- Bare ground, litter cover, family richness, and plant functional types (particularly the presence of shrubs and tree seedlings) are useful indicators of tracked vehicle effects on understory vegetation
- Certain species of concern should still be monitored actively
- These indicators nest into the broader suite of indicators across spatial scales across Fort Benning



## Acknowledgements:

- **Financial support: Strategic Environmental Research & Development program (SERDP)**
- **SERDP Support: Hal Balbach, Hugh Westbury, & Robert Holst**
- **Bulldozer operators: Troy Key & Phil Bennet**
- **Field assistants: Keiran O'Hara, Tom Govus, James Cantu, & Jennifer Ayers**
- **Additional Photos: Kelly Maloney & James Cantu**





## Technology Transfer of Ecological Indicators:

### Characterization of Microbial Communities as Indicators of Land Usage at Fort Benning, GA

Center for Biomarker Analysis



Soil biota are useful measures of the functional integrity of terrestrial ecosystems. That is, soil microorganisms serve as indicators of the nutrient status and of the ongoing ecological processes that can potentially predict the future status of an ecological system. We field tested a methodology at Fort Benning that measures shifts in microbial biomass, community composition, and physiological status (measured as signature lipid/DNA biomarkers) as a quick and easy indicator of changes in soil quality as a result of military activities and land management.

Managers at military installations provide land for training of military personnel. Often, such activities are inconsistent with sustainable land use practices. An effective monitoring program capable of predicting recovery from training use is essential to ensuring the long-term viability of these lands. Previously, the majority of these monitoring programs have focused on biodiversity of terrestrial macroorganisms that have a long period of recovery and require expensive monitoring specialists. Microorganisms that can be quantitatively monitored with chemical biomarkers have been largely overlooked despite their complete integration and dependency with the macro world (Zak et al., 1994; Lee, 1991).

Peacock, A. D., S. J. MacNaughton, J. M. Cantu, V. H. Dale and D. C. White. 2001. Soil microbial biomass and community composition along an anthropogenic disturbance gradient within a longleaf pine habitat. *Ecological Indicators* 1(2):113-121.





## Characterization of Microbial Communities



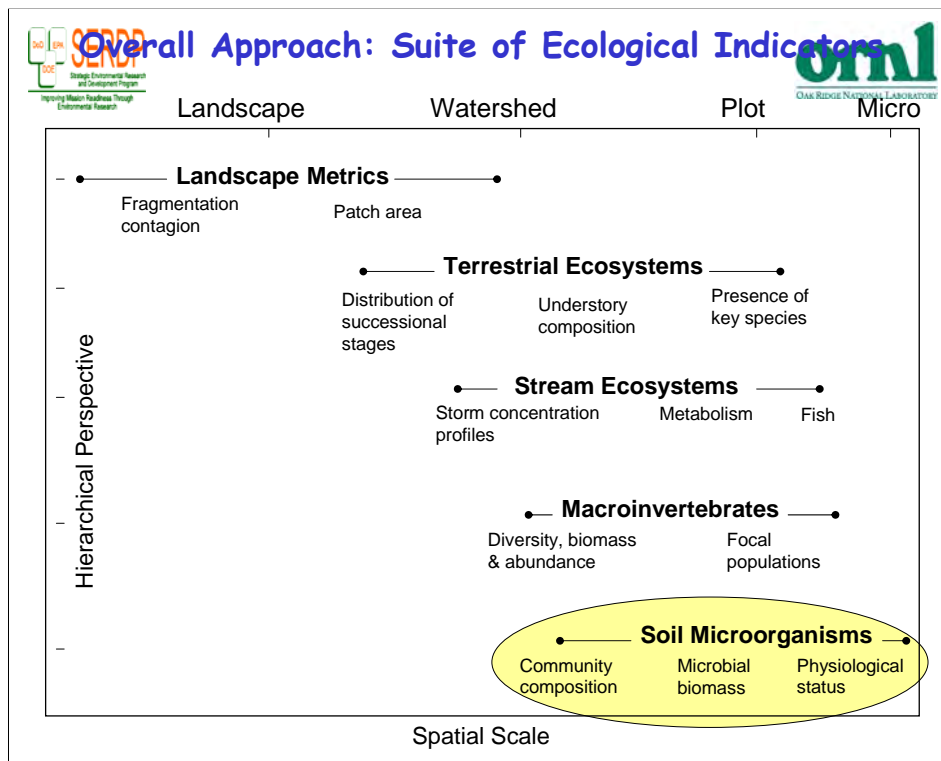
### Principal Investigators:

- Aaron Peacock, The University of Tennessee
- James Cantu, The University of Tennessee
- Virginia Dale, Oak Ridge National Laboratory
- David White, The University of Tennessee
- A. Smithgall, The University of Tennessee

*Center for Biomarker Analysis*







The response of microbial biota is part of a larger project designed to investigate indicators that would be useful to augment current sampling regimes at military bases and typical of other actively managed sites.



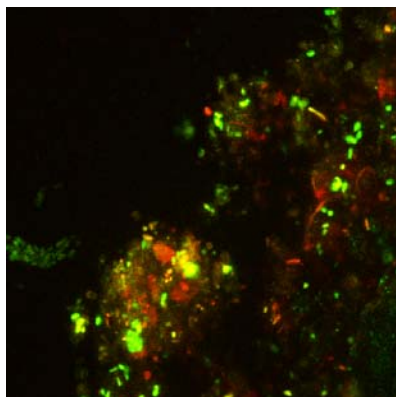
- What can the microbial community tell an investigator about land use?
- What tools can be used to assess microbial communities in-situ?

No single property is a sole indicator of soil quality, but the collection and analysis of many accepted soil quality indicators can be time-consuming and costly. Biological indicators of the changes of *in situ* microbial ecology, such as viable biomass, community composition, and physiological status, have been shown to correlate well to changes in other soil quality indicators (i.e., soil porosity, texture, organic matter, phosphorus concentrations, nutrient cycling, and concentrations of elements). Because changes in these microbial indicators reflect changes in other geophysical properties of the environment, they may serve as a good overall indicator of soil quality. Therefore, a methodology that uses these biological indicators would be a useful tool for land managers to cost-effectively determine changes in soil quality that may result from human use of the site.

We therefore investigated the potential use of signature biomarker analysis for early detection of changes in soil quality as well as soil quality itself and vegetation indicators that change as a result of military land use and land management activities. We analyzed the microbial community response to ecological change by employing lipid biomarker analyses, specifically phospholipid fatty acid analysis (PLFA). PLFA analysis can determine community structure shifts within the whole viable microbial biomass in response to ecological change. The approach of PLFA analysis has been used in habitats ranging from contaminated groundwater through subsurface and surface soils (Stephen et al., 1999; MacNaughton et al., 1999) and has provided a comprehensive and detailed picture of changes in microbial community structure.



## Answers

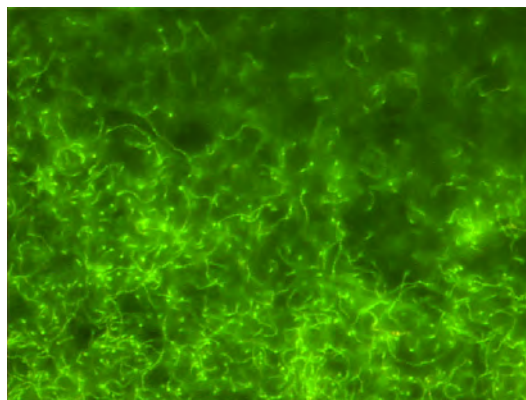


Microbial biomass can react quickly to conditions of nutrients, moisture, temperature, and the type and amount of organic matter levels (Paul, 1984).

Microbial biomass in soil has a turnover time of less than one year and as such reacts quickly to conditions that change the type and amount of soil organic matter levels (Paul, 1984).



## Answers (Continued)

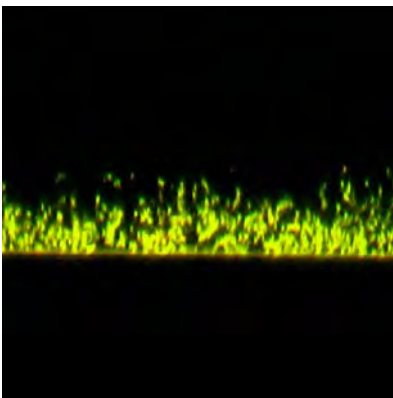


Viable microbial biomass is integral for nutrient storage and cycling, filtering, buffering, and gene reserves (Blum, 1998).

Additionally microbial biomass is the living part of the soil organic matter and is responsible for nutrient storage and cycling (Rice et al. 1996), soil aggregate formation (Tisdale and Oades, 1982), and other ecological uses such as filtering, buffering and gene reserves (Blum, 1998).



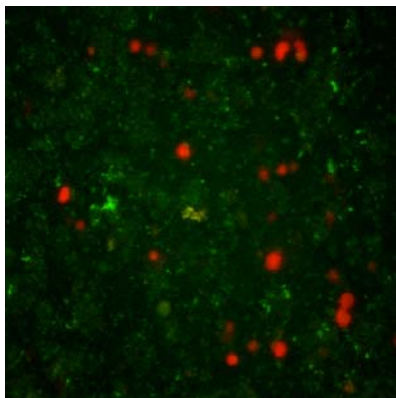
## Answers (Continued)



Microbial biomass and community composition have been shown to be sensitive indicators of changes in nutrient type, botanical composition, pollutant toxicity, and climate change



## Answers (Continued)

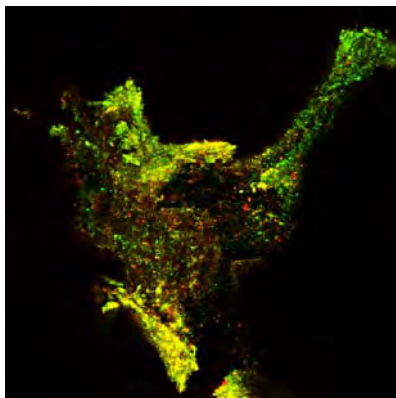


Because the microbial community integrates the physical and chemical aspects of a site and responds to anthropogenic activities, it can be considered a biological indicator.

Soil microbial biomass and community composition has been shown to be a sensitive indicator of changes in nutrient type (Peacock et al. 2001, Kirchner et al. 1993), botanical composition (Borga et al. 1994), pollutant toxicity (Stephen et al. 1999), and climate change (Zogg et al. 1997). Because the microbial community integrates the physical and chemical aspects of the soil and responds to anthropogenic activities, it can be considered a biological indicator of soil quality (Rice et al. 1996).



## Answers (Continued)



The microbial viable biomass, community composition, and nutritional status can be readily measured to provide rational endpoints for many disturbance/recovery processes  
(White et. al., 1998)

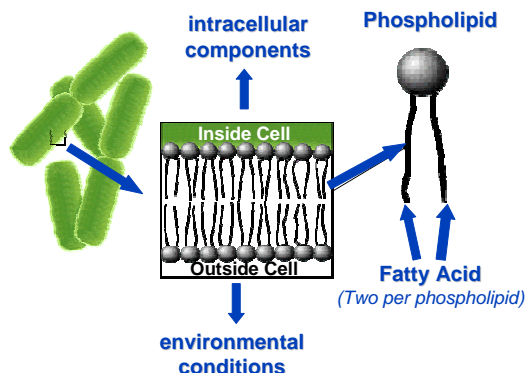


- Imagine having several Billion sensors the size of a few microns at a contaminated site.
- This is essentially the case when analyzing the in situ microbial community.



# What are phospholipids?

PLFA profiles reflect both the nature of the intracellular components and the extracellular environmental conditions.



PLFA analysis tells us **what types of microbes are present** in a system and **how they are reacting** to environmental factors (pollution, disturbances etc.).

## PLFA Analysis

Solvents were GC grade from Fisher Scientific, Fairlawn, NJ. Each lot was tested for purity before use. Silicic acid was Unisil (Clarkson Chemical Company). Potassium hydroxide and the internal fatty acid standard 19:0 were from Sigma Chemical Co., St. Louis, MO. Glassware for lipid analysis was freed of lipid contaminants by washing with non-phosphate containing soap and then heating in a muffle furnace at 450°C for 4 hours.

Soil was extracted with the single-phase chloroform-methanol-buffer system of Bligh & Dyer (1954), as modified (White et al., 1979). To 5 g soil samples, 5 mL of chloroform, 10 mL of methanol, and 4 mL of phosphate buffer (50 mM, pH = 7.4) were added, mixed, and allowed to equilibrate for 3 hours. The single phase extractant was separated from the solid material by centrifugation at 2000 rpm for 20 minutes and decanting into another test tube. Five mL of chloroform was used to wash the pelleted solids, which were then re-centrifuged, and the chloroform added to the extract. An additional 5 mL of water was added to the extract to force the separation of the aqueous phase from the organic phases. After separation for approximately 12 hours, the organic phase was pipetted into a new test tube and the solvent removed with a stream of dry nitrogen at 37°C.

The total lipid extract was fractionated into neutral lipids, glycolipids, and polar lipids by silicic acid column chromatography (Guckert, et al., 1985). Pasteur pipets (1 cm diameter) partially blocked with a plug of glasswool were prepared and 0.5 g silicic acid added as a slurry in 0.02 M ammonium acetate in methanol. The columns were pre-eluted with 5 mL of acetone, 5 mL of chloroform and sample transferred to the column with 3-100 µL washes of chloroform. Neutral lipids were eluted with 5 mL chloroform, glycolipids with 5 mL acetone, and polar lipids with 10 mL of methanol. The solvent was removed from the polar lipids under a stream of dry nitrogen at 37°C. All results presented in this paper are for the polar lipid fraction.

The polar lipids were transesterified to the fatty acid methyl esters by a mild alkaline methanolysis (Guckert, et al., 1985). The polar lipid extract was dissolved in 1 mL of 1:1 chloroform/methanol, then 1 mL of methanolic KOH was added, and the mixture was heated at 60°C for 30 minutes. Fatty acids methyl esters (FAMES), were recovered from the organic fraction of the sample after adding 2 mL of hexane and 2 mL of water to break phase.

The FAMES were analyzed by capillary gas chromatography with flame ionization detection on a Hewlett-Packard 5890 Series 2 chromatograph with a 50 m non-polar column (0.2 mm I.D., 0.11 µm film thickness). The injector and detector were maintained at 270°C and 290°C, respectively. The column temperature was programmed from 60°C for 2 minutes then ramped at 10°C min<sup>-1</sup> to 150°C, then ramped to 312°C at 30°C min<sup>-1</sup>. Preliminary peak identification was by comparison of retention times with known standards.

Definitive identification of peaks was by gas chromatography/mass spectroscopy of selected samples using a Hewlett-Packard 6890 series gas chromatograph interfaced to a Hewlett-Packard 5973 mass selective detector using a 20 m non-polar column (0.1 mm I.D., 0.1 µm film thickness). The injector and detector were maintained at 290°C and 300°C respectively. The column temperature was programmed from 60°C for 1 minute then ramped at 20°C min<sup>-1</sup> to 150°C, held for 4 minutes, then ramped at 70°C min<sup>-1</sup> to 230°C held for 2 minutes, and finally ramped at 10°C min<sup>-1</sup> and held for 3 minutes. Mass spectra were determined by electron impact at 70 eV. Methyl nonadecanate was used as the internal standard, and the PLFA expressed as equivalent peak response to the internal standard.



**PLFA analysis provides quantitative insights into three important attributes of microbial communities:**

- **Viable Biomass**  
*(how many cells are present)*
- **Community Structure**  
*(what types of microbes are there)*
- **Metabolic Activity / Physiological Status**  
*(what are they doing: active – slow – adapting)*



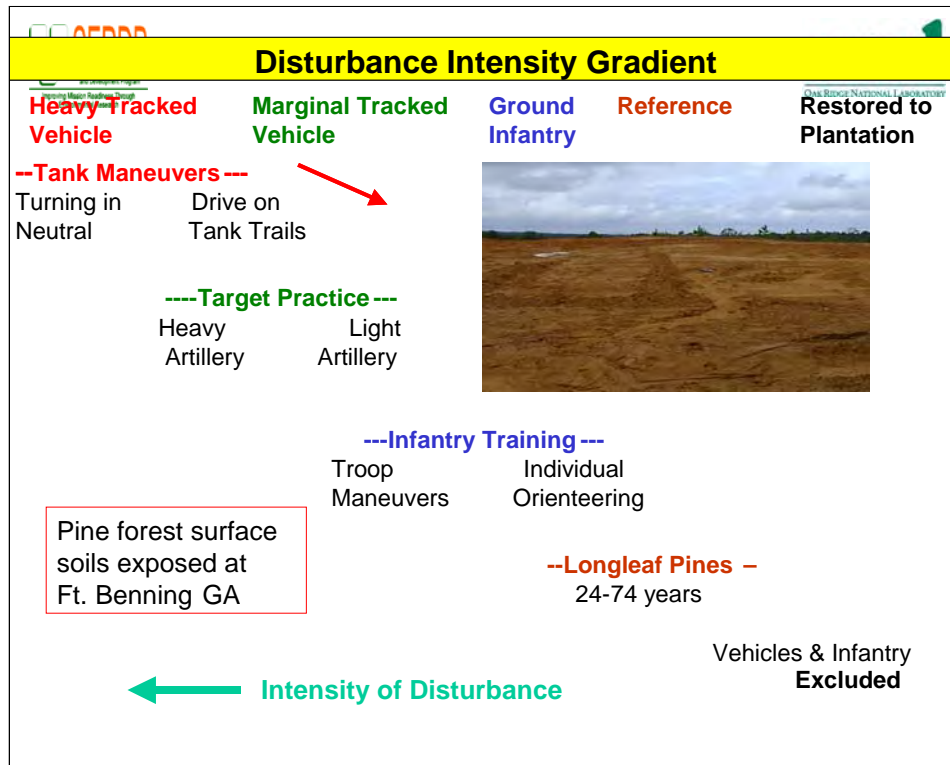
## Disturbance Gradient Fort Benning, GA



We investigated the utility of using the soil microbial biomass and community composition as ecological indicators of change along an anthropogenic disturbance gradient. The disturbance gradient included the duration and type of traffic in military training areas in a longleaf pine habitat. The hypothesis was that duration and intensity of disturbance (traffic) in the longleaf pine ecosystem would be reflected in changes in the soil microbial community biomass and structure. These changes could be quantitatively measured by phospholipid ester-linked fatty acid analysis (PLFA). In addition, we used two different data analysis techniques to classify disturbance, the first, a linear discriminant analysis classified transects based on 17 PLFA variables, the second a non-linear artificial neural network analysis which included all 61 PLFA variables and the biomass in which to base predictions. Herein we compare these two analyses.

Disturbance of the soil ecosystem due to training includes the direct removal or damage of terrestrial vegetation, digging activities, dislocation, and compaction from vehicles, erosion, and sedimentation. The degree and extent of the impacts of training activities within a compartment are dependent upon the type of activity, number of personnel training, and how frequently the compartment is exposed to activity. Furthermore, training activity typically occurs irregularly throughout a compartment, creating localized gradients of disturbance within individual compartments.





In this study we investigated the utility of using the soil microbial biomass and community structure as ecological indicators of change along an anthropogenic disturbance gradient, in this case the amount and type of traffic in military training areas in a longleaf pine habitat. We hypothesized that disturbance in the longleaf pine ecosystem from low to high levels in the form of traffic would induce changes in the soil microbial community biomass and structure and would be measurable by phospholipid fatty acid analysis.



## Sample collection

- Choose a representative area of land use
- Soil samples are collected with push cores
  - Identifiable Organic Matter is removed and the mineral “A” horizon is sampled (approximate depth of 5 cm)
  - Eject soil from push core into a sample bag
  - Store on blue ice
  - Ship overnight to laboratory



Samples can be collected from soil cores. To avoid cross contamination in between each sample, the soil cores should be washed in solvent (methanol) and sterile distilled water. Our cores were approximately 20 cm in length and 2 cm in width. For each core taken, the depth of the core and the presence/absence of an “A” horizon should be measured. For our study, five samples were taken from separate plots at each transect (14 transects x 5=70 samples, as part of the observational study mentioned previously). Of the transects selected, three were reference transects (with stand ages 28, 68, and 74 years); three were heavy usage (undergoing tracked vehicle training); three were moderate usage (areas adjacent to tracked vehicle training); three were light usage (infantry training), and two came from a site currently undergoing remediation (previous heavy disturbance, currently trees and groundcover planted and no usage). Samples were stored at  $-80^{\circ}\text{C}$  prior to analysis.



## Soil Chemical Biomarker Measurements For Quantitative Disturbance Impacts & End Points for Reversibility

### Predictive Analysis of disturbance using the soil microbial community

- **TWO APPROACHES:**
- Linear Discriminant model using 17 PLFA predictor variables
- **Two groups** ↑ disturbance ↑ in actinomycetes & spore-former Gram positive bacteria, ↓ in gram-negative bacteria and microeukaryotes
- Non-linear Artificial Neural Network Analysis using all 60 PLFAs and microbial biomass
- Predict classification 66% of time (Chance = 25%)  
Hysteresis in recovery from sensitivity

#### *Statistical Analysis*

The relative proportion (percentage mole fraction) or biomass (pmol/g) of PLFA was used to test the null hypothesis that degree of land disturbance would not influence the composition of the soil microbial communities. To test that hypothesis, an analysis of variance (ANOVA) using the General Linear Model STATISTICA procedure (Statsoft Inc. Tulsa, OK) for a completely randomized design with five treatments was used. The values reported are least square means of 15 replicates, except in the case of the remediated treatment which contained 10 replicates (total n=70) standard errors of the means were determined. Differences in the mean proportions of PLFA in each treatment were tested using Tukey's Honest-Significant-Difference procedure. A hierarchical cluster analysis (Ward's method, 1-Pearson r) was used to discover how the PLFAs which differed significantly with treatment clustered.

A discriminant analysis with cross-validation was chosen to classify the observations into one of 4 classes (n=60, 15 observations in each group) based on the degree of land disturbance. Only those PLFA that comprised at least 1% of any profile were included in the analysis; therefore fatty acids that may have been unreliably quantified were not included. Before statistical analysis arcsine square root transformation was applied to the mole percent PLFA data. After truncation, a one-way ANOVA was conducted on the remaining PLFAs' and those that differed significantly with usage were included in the model.

#### *Artificial Neural Network Analysis*

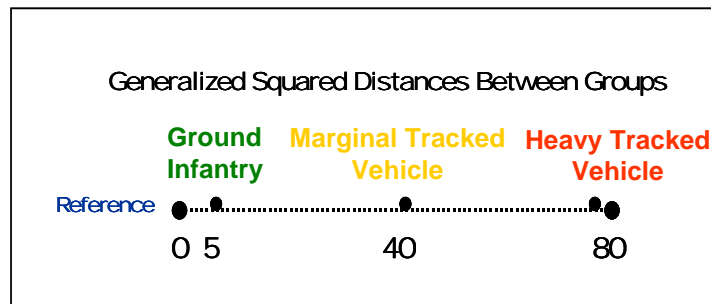
Neural network identification was performed with early stopping by cross-validation and topology optimization by bootstrapping (selection criteria: median cross-validated error) using microCortex web based neural computing environment ([www.microcortex.com](http://www.microcortex.com)).



Linear Discriminant analysis showed that the reference and light transects were very similar while the moderate and heavy transects greatly differed in regards to the microbial community structure.

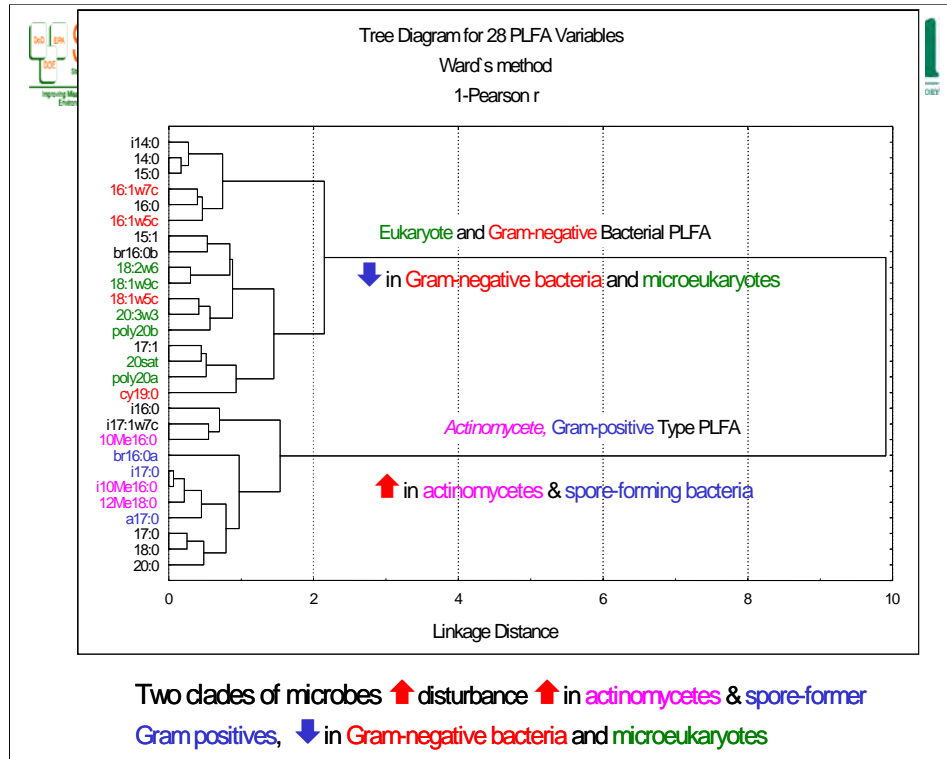
PLFAs included in the Discriminant Model

|          |           |          |
|----------|-----------|----------|
| a15:0    | i17:0     | 18:1w9c  |
| i16:0    | a17:0     | 18:0     |
| 16:1w7c  | Cy17:0    | 10Me18:0 |
| br16:0a  | 17:0      | Cy19:0   |
| i17:1w7c | i10Me16:0 | 20sat    |
| 10Me16:0 | 18:2w6    |          |

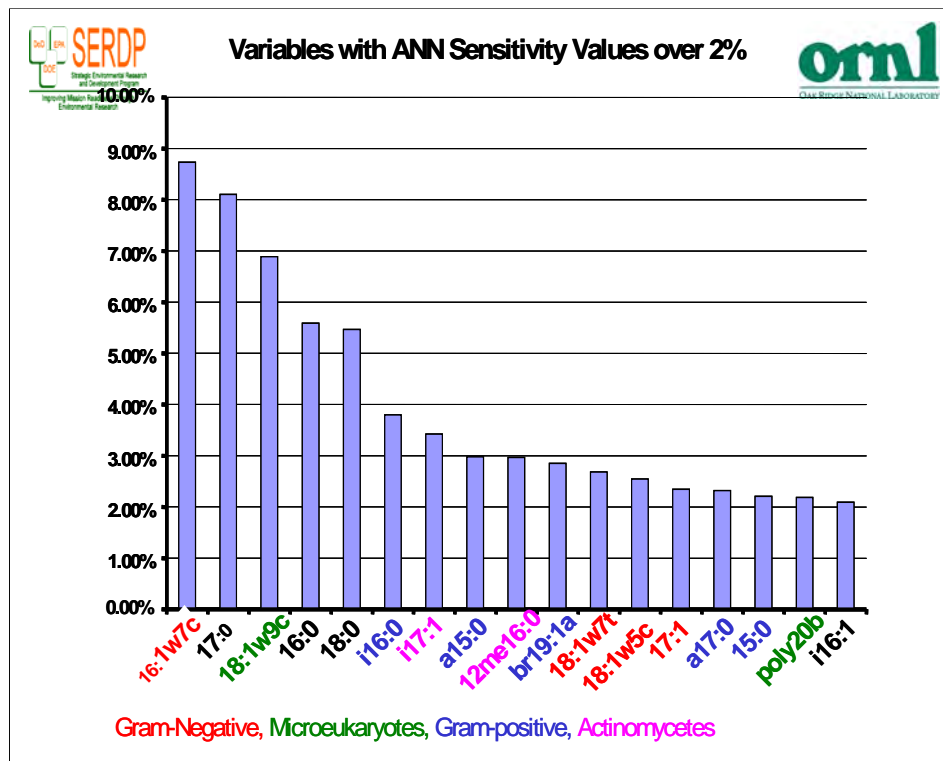


A discriminant analysis with cross-validation was chosen to classify the observations into one of 4 classes (n=60, 15 observations in each group) based on the degree of land usage. The first task was to reduce the number of variables to be included in the model. Only those PLFA that comprised at least 1% of any profile were included in the analysis; therefore fatty acids that may have been unreliably quantified were not included. Before statistical analysis arcsine square root transformation was applied to the mole percent PLFA data. After truncation, a one-way ANOVA was conducted on the remaining PLFAs' and those that differed significantly with usage were included in the model. The resulting model included 17 descriptor variables. Wilks' Lambda for the model was .032 ( $P < .001$ ). Overall the Error estimates for the model were 33% and the generalized distance between groups is reported. Only the first 4 treatments were used to construct the model, once the model was complete the 10 observations taken from the remediated transects were classified. One observation was classified as a reference, three as light trafficked, and 6 as moderately trafficked.









A non-linear Artificial Neural Network analysis (ANN) was performed using the biomass estimates and all of the 61 PLFA variables. The resulting ANN included 5 hidden nodes and resulted in an  $r^2$  of 0.97. The hit rate for this model was 66% and 6 of the PLFAs had sensitivity values above 3%. As with the discriminant model, once the ANN model was complete it was used to classify the observations from the remediated transects. Four of the observations were classified as reference, two as moderate, and four as heavily trafficked.





## Laboratories Capable of PLFA Analysis



- **The University of Tennessee Center for Biomarker Analysis**  
<http://cba.bio.utk.edu>
- **Microbial Insights Inc.**  
[www.microbe.com](http://www.microbe.com)



## Data Analysis

- [www.microcortex.com](http://www.microcortex.com) (ANN)
- The University of Tennessee Center  
for Biomarker Analysis  
<http://cba.bio.utk.edu> (Discriminant)
- Data is stored in the SEMP data  
repository



## Conclusions

The analysis based on the soil PLFA was successful, reflected above-ground changes, and provided an index of the degree of land disturbance (traffic) the soil received. Both linear discriminant and non-linear ANN analysis were able to adequately classify the degree of disturbance.

The goal of this project was to explore the possibility of using the soil microbial community as an indicator of the degree of soil degradation along a disturbance gradient. We hypothesized that the soil microbial community would reflect above-ground changes and provide an indication of the degree of land disturbance, in this case the amount of traffic the soil received. Using a combination of linear and non-linear analyses we were able to distinguish the impact of traffic along a disturbance gradient with an appropriate degree of accuracy. This knowledge provides land managers with a valuable tool to aid in the assessment of soil disturbance in these areas.



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Journal (published or in press): 14  
 Journal (in review or submitted): 3  
 Journal (in preparation): 4  
 Book chapters or symposium proceedings: 2  
 Master's Thesis: 1  
 Dissertations: 3

## **Presentations:**

Professional meetings: 30 (4 presentations in symposia and 4 as plenary lectures)  
 Other meetings: 8

## **Award:**

Kelly Maloney was awarded The Carolyn Taylor Carr Outstanding Award Dissertation for 2004-2005 from the Auburn Chapter of Sigma Xi

Kelly Maloney was awarded Auburn's Department of Biological Sciences Outstanding Graduate Research Award.

## **Published or in press papers**

Black, B.A., H. T. Foster, and M.D. Abrams. 2002. Combining environmentally dependent and independent analysis of witness tree data in east-central Alabama. **Canadian Journal of Forest Research** 32: 2060-2075.

Dale, V.H. and Beyeler, S.C. 2001. Challenges in the development and use of ecological indicators **Ecological Indicators** 1: 3-10.

Dale, V.H., Beyeler, S.C., and Jackson, B. 2002. Understory indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. **Ecological Indicators** 1(3): 155-170.

Dale, V. H., Mulholland, P., Olsen, L. M., Feminella, J., Maloney, K., White, D. C., Peacock, A., and Foster, T. 2004. "Selecting a Suite of Ecological Indicators for Resource Management," Pages 3-17 in *Landscape Ecology and Wildlife Habitat Evaluation: Critical Information for Ecological Risk Assessment, Land-Use Management Activities and Biodiversity Enhancement Practices*, **ASTM STP 11813**, L. A. Kapustka, H. Gilbraith, M. Luxon, and G. R. Biddinger, Eds., ASTM International, West Conshohocken, PA, 2004.

Dale, V. H., S. Archer, M. Chang, and D. Ojima. 2005. Ecological impacts and mitigation strategies for rural land management. *Ecological Applications*. **Ecological Applications** 15(6): 1879-1892.

Dale, V.H., Druckenbrod, D., Baskaran, L., Aldridge, M., Berry, M., Garten, C., Olsen, L., Efroymsen, R., and Washington-Allen, R. 2005. Vehicle impacts on the environment at different spatial scales: Observations in west central Georgia. **Journal of Terramechanics** 42:383-402.

Dale, V.H., D. Druckenbrod, L. Baskaran, C. Garten, L. Olsen, R. Efroymsen, and R. Washington-Allen, M. Aldridge, M. Berry. 2005. Analyzing Land-Use Change at Different Scales in Central Georgia. Pages 1-4 in *Proceedings of the 4th Southern Forestry and Natural Resource GIS conference*. Athens, Georgia, Dec 16-18.

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Houser, J.N., Mulholland, P.J., and K. Maloney. In press. Stream chemistry indicators of disturbance on military reservations. **Journal of Environmental Quality**



Maloney, K.O., P.J. Mulholland, and J.W. Feminella. 2005. Influence of catchment-scale military land use on physicochemical conditions in small Southeastern Plains streams (USA). **Environmental Management** 35(5): 677-691.

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Maloney, K.O., Richard M. Mitchell and J.W. Feminella. In press. Influence of catchment disturbance on fish integrity in low-diversity headwater streams.

Mulholland, P. J., J. N. Houser, and K. O. Maloney. 2005. Stream diurnal dissolved oxygen profiles as indicators of in-stream metabolism and disturbance effects: Fort Benning as a case study. **Ecological Indicators** 5: 243-252.

Olsen, L.M., Washington-Allen, R.A, and V.H. Dale. 2005. Time-series analysis of land cover using landscape metrics. **GIScience and Remote Sensing** 42(3): 200-223.

Olsen, L.M., Dale, V.H. and H.T. Foster. In press. Landscape patterns as indicators of ecological change at Fort Benning, GA. **Land Use and Urban Planning**.

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#### **Submitted Papers:**

Wolfe, A., and Dale, V.H. A. Ecological research for natural resource management: Using a Delphi approach to integrate science and practice.

Date of submission: May 2004

Journal: **Journal of Environmental Management**

Dale, V.H., Peacock, A., C. Garten, and E. Sobek. Contributions of soil, microbial, and plant indicators to land management of Georgia pine forests.

Date of Submission; November 2005

Journal: **Ecological Indicators**

#### **In Preparation:**

Dale, V.H., Baskaran, L., Cantu, J., Druckenbrod, D.L., Feminella, J., Maloney, K., Mulholland, P., and A. Peacock (Second authors listed alphabetically). Synthesis of ecological indicators research across multiple scales at Fort Benning, Georgia.

Journal: **BioScience**

Druckenbrod, D.L., and V.H. Dale. Experimental response of understory plants to mechanized disturbance in an oak-pine forest.

Date of submission: March 2006

Journal: **Applied Vegetation Science**

Druckenbrod, D.L., Ayers, P., Cantu, J., Dale, V.H., Feminella, J., Garten, C., Maloney, K., and A. Peacock (Second authors listed alphabetically). Experimental response of ecological indicators to mechanized disturbance in longleaf pine forests.

Journal: **Ecosystems**

Maloney, K. O, J. W. Feminella, P. J. Mulholland, S. A. Miller, and J. N. Houser. Land use legacies and small streams: the relationships between historic and contemporary land use patterns with present-day stream conditions.



Journal: ?**Ecosystems**?

**Dissertations and Thesis:**

Foster, H.T., II. 2001. Long term average rate maximization of Creek Indian residential mobility a test of the marginal value theorem. Ph.D. dissertation, Department of Anthropology, Pennsylvania State University.

Beyeler, S.C. 2000. **Ecological indicators**. Master's thesis. University of Miami in Ohio.

Maloney, K. 2004. The influence of catchment-scale disturbance on low-order streams at Fort Benning, Georgia, USA. Ph.D. dissertation. Auburn University, Alabama.

Peacock, A. Ecological Indicator Development, Integration and Knowledge Mapping" Ph.D. Dissertation, The University of Tennessee Department of Biosystems Engineering.  
Anticipated: 2005